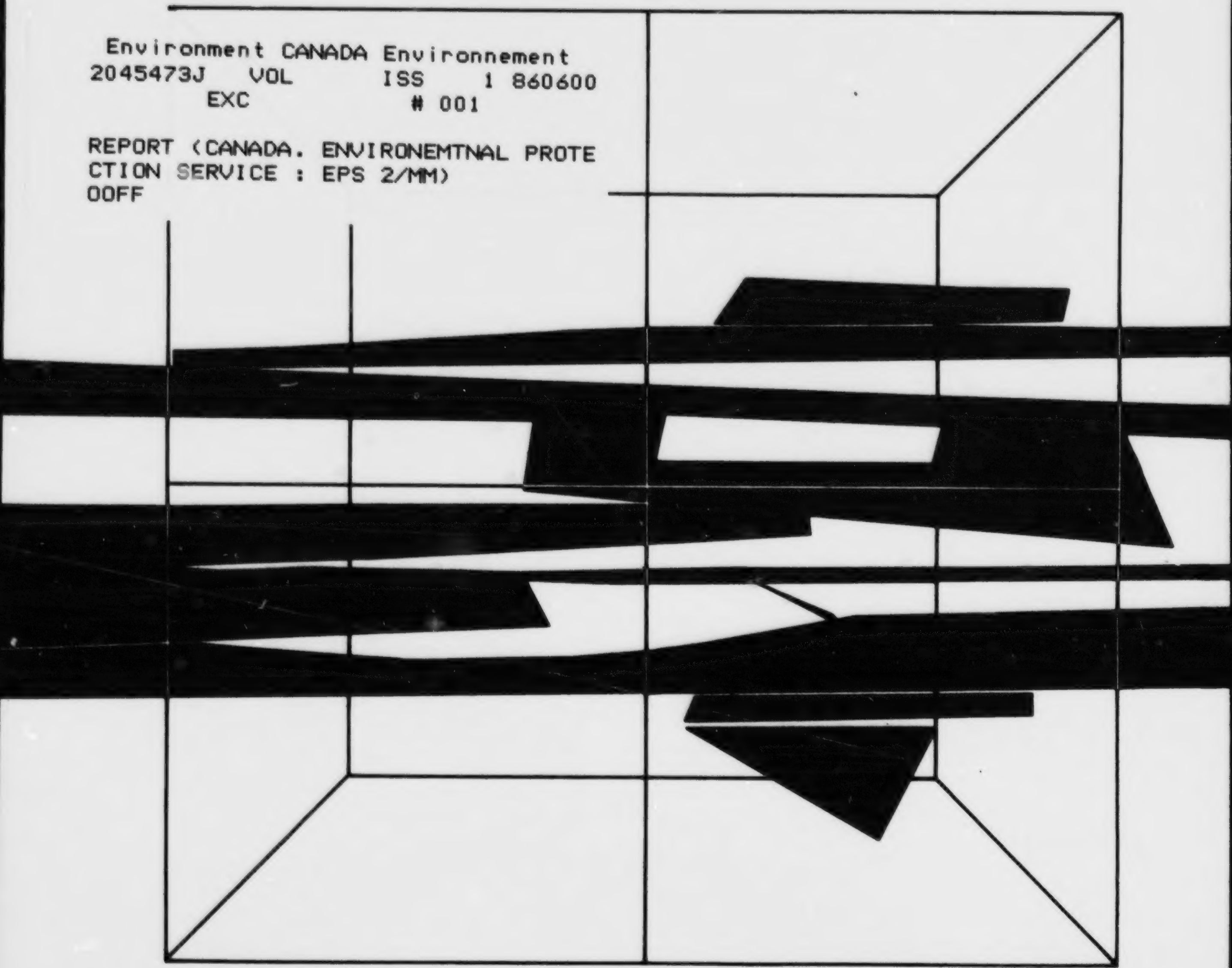
An Assessment of the Radiological Impact of Uranium Mining in Northern Saskatchewan

Report EPS 2/MM/1
June 1986



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AN ASSESSMENT OF THE RADIOLOGICAL IMPACT OF URANIUM MINING IN NORTHERN SASKATCHEWAN

prepared for

Environment Canada
and the
Atomic Energy Control Board

by

IEC Beak Consultants Ltd.

Report EPS 2/MM/1
June 1986

REVIEW NOTICE

This report has been reviewed by the Conservation and Protection Service, Environment Canada, and the Atomic Energy Control Board and approved for publication. Approval does not necessarily signify that the contents reflect the views and policies of Environment Canada or the Atomic Energy Control Board. Mention of trade names or commercial products does not constitute recommendation or endorsement for use.

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ABSTRACT

This report presents the findings of a study which investigated the regional radiological impact of uranium mining in northern Saskatchewan. The study was performed by IEC Beak Consultants Ltd. under a contract awarded by Environment Canada in partnership with the Atomic Energy Control Board.

This preliminary assessment suggests there is a negligible combined regional radiological impact from simultaneous operation of the three operating mines investigated as part of the present study. The mines are spaced too far apart for any superposition of emissions to be significantly greater than a small fraction of background levels.

The most exposed individual not directly associated with any of the mining operations is estimated to receive a total radiation dose equal to about 3% of the dose due to natural background radiations. This increment is equivalent to the increment in natural background that would be received by an individual moving from Vancouver to Wollaston Post, before mining began in the area, as a result of reduced atmospheric shielding from cosmic radiation.

Radiological impacts on biota are estimated to have insignificant effects on natural populations in all cases. However, since the study only investigates the effects of operational releases of radionuclides, the results do not imply that uranium mining developments will or will not have significant long-term radiological impact on northern Saskatchewan.

Radiological impact assessments described in this report are estimates only. There are some uncertainties in the available data and modelling methodology. The radiological impact of abandoned tailings areas was not included in this study.

RÉSUMÉ

Le rapport présente les résultats d'une étude sur les effets régionaux des rayonnements ionisants dus à l'extraction de l'uranium dans le nord de la Saskatchewan. Commandée par Environnement Canada, de concert avec la Commission de contrôle de l'énergie atomique, cette étude a été réalisée par IEC Beak Consultants Ltd.

Cette évaluation préliminaire semble indiquer que l'exploitation simultanée des trois mines sur lesquelles l'étude a porté a eu un effet régional cumulatif négligeable. Les mines étant suffisamment distantes les unes des autres, l'effet de la superposition des rayonnements émis ne peut être beaucoup plus considérable qu'une petite fraction du fond de rayonnement.

On estime que l'individu le plus exposé qui n'est en rapport direct avec aucune des opérations minières reçoit une dose totale équivalente à environ 3% de la dose due au rayonnement naturel. Une personne qui serait partie de Vancouver pour se rendre à Wollaston Post, avant le début de l'exploitation minière dans cette région, aurait reçu une dose semblable en raison de la protection réduite de l'atmosphère contre le rayonnement cosmique.

Dans tous les cas, on estime négligeable l'effet des rayonnements sur les populations naturelles de flore et de faune. Toutefois, comme l'étude ne porte que sur l'effet des émissions de radionucléides pendant la période d'exploitation, les résultats n'indiquent pas si l'exploitation des mines d'uranium aura ou non d'importantes conséquences à long terme sur le nord de la Saskatchewan en ce qui concerne les rayonnements.

Les évaluations présentées dans le rapport ne sont qu'approximatives. Les données disponibles et les méthodes employées comportent quelques incertitudes. L'effet des rayonnements provenant des résidus de traitement des minerais d'uranium n'a pas été étudié.

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SUMMARY AND RECOMMENDATIONS

S.1 Overview of Assessment Methods

The purpose of this study is threefold: (1) to assess the regional radiological impact of uranium mining development in Saskatchewan, (2) to provide some appreciation of the magnitude of radiological doses to individuals and to biota, and (3) to illustrate an assessment methodology.

This report is an update of a draft report prepared in 1980 for the Environmental Protection Service, Environment Canada, which considered five mining operation sites in northern Saskatchewan. However, in this study, only three operational mining facilities are considered. Recent changes in mining operations, including the close-out of Beaverlodge and the postponement of the Midwest Project, required revisions to the report to ensure it is representative of existing conditions. IEC BEAK Consultants has taken advantage of this review to include the more modern and accurate assessment methods which have become available since the 1980 report.

This study was intended to provide a regional overview based upon the best available information at the time of writing. Use of any of the modelling information generated in this study for estimating radiological impacts in the immediate proximity of the facilities should be discouraged. The present models cannot be applied beyond the intended context.

The population data utilized in this study is for 1981, the latest census data available. All emissions data are based on expected releases from mining and milling when each facility has reached mature operation predicted for sometime in the mid-1980's. Abandoned tailings and waste rock areas were not considered as source terms for regional modelling.

Source Terms. A review of uranium mining developments in northern Saskatchewan, and mining, milling, and waste management practices was carried out. Quantitative estimates of air emissions and liquid discharges of radioactive materials were then made for all three identified mine sites currently operating. These mine sites are:

1) Rabbit Lake (Eldor Resources)
2) Cluff Lake (Amok)
3) Key Lake (Key Lake Mining Company)

The quantitative estimates of air emissions and liquid discharges are called "Source Terms". Source term estimates were based on data provided by mine operators and their consultants; by the Environmental Protection Service, Environment Canada; Saskatchewan Environment; and supplemented by literature information, primarily from U.S. sources. The tailings resuspension term was derived by running the MILDOS computer program. A deliberately wide range of source terms was chosen to illustrate a potential range of effects. This study considers only the radionuclides which are in the uranium decay series.

Source terms may be subject to review at a later date as the mining companies gain more operational experience. Changes in source terms may alter the estimated radiation doses arising from any operation.

Environment. A review of the climatology and of the terrestrial and aquatic environments in northern Saskatchewan was carried out. Drainage areas and streamflow data were compiled and used in developing chain-lake-river models for the Lake Athabasca and Reindeer Lake drainage basins. The food web in northern Saskatchewan was reviewed, and important terrestrial and aquatic biota species were identified for subsequent quantitative assessment. Human uses of the environment were also considered. These data form the basis for most of the dose assessment modelling for both natural and human populations.

Atmospheric Dispersion of airborne emissions, ground deposition, and resuspension of radioactive materials were calculated using the MILDOS computer code. Subsequent transfers to vegetation and animals were calculated by models, using either site-specific parameter values or selected literature values after careful review of the data. These models were supplemented by others developed for the sub-arctic food-chain: air-to-lichens-to-caribou-to-man.

Radiation Doses to Individuals from external and internal pathways were calculated in detail for only one site (the Eldor Resources Operation). Since this facility has the highest population density, it was considered sufficient to illustrate the magnitude of the doses and the assessment methodology. Collective doses, however, were calculated for all populations within 80 km of all three mine sites and for those along affected drainage basins. All internal dosimetry calculations were based on methods of the International Commission on Radiological Protection (ICRP 26 1977), using the committed effective dose equivalent approach. Both inhalation and ingestion components were calculated for internal doses.

Radiation Doses to Selected Terrestrial Biota. These were mainly calculated using methodology recommended by the International Atomic Energy Agency (IAEA), Vienna; some of the parameter values used in the dosimetry models for zooplankton were developed in this study. These doses shall generally be regarded as upper limit estimates.

Aquatic Dispersion. Dispersion of discharged radioactive materials in the watersheds of northern Saskatchewan was calculated using a chain-lake-river model. The model assumes uniform mixing in any given lake and is intended to predict long-term concentration trends in the watersheds rather than short-term variations. It allows for radioactive decay, removal of radioactive materials to sediments, and diffusion of radioactive materials from sediments.

Transfer of radioactive materials to aquatic biota such as fish, fish eggs, invertebrates, and aquatic plants was calculated using bioaccumulation factors which relate the concentrations of radioactive materials in a given organism to that in the surrounding water. Bioaccumulation factors were selected after detailed review of the published data.

S.2 Doses to Individuals

Detailed calculations of doses to individuals were made for the Wollaston Lake community. This is the only community of any size within 80 km of any of the three mining sites. Wollaston Lake has a population of 476 and is 33 km from the Rabbit Lake site. For dose calculation purposes, the Wollaston Lake community is considered the critical group. Residents are assumed to live there year round, and to take all drinking and bathing water from Wollaston Lake. They are assumed to eat moose and caribou (50 kg/a) shot in the area, to consume fish (44 kg/a) from Wollaston Lake, and to eat locally grown vegetables (50 kg/a). Calculated doses to individuals (based on ICRP 26 methodology) from both aquatic and atmospheric pathways are listed in Table S.1.

The total effective dose equivalent received by the most exposed member of the critical group was estimated to be 35 μSv/a. The estimated internal dose equivalent accounted for 34 μSv/a or 94% of the total.

The external dose component includes doses from immersion in air and from exposure to contaminated ground, water, and beaches. The health effects of skin dose are minimal since skin is not very radiosensitive. Doses from the inhalation of radon

TABLE S.1 ESTIMATED ANNUAL EFFECTIVE DOSE EQUIVALENTS TO MOST HIGHLY EXPOSED INDIVIDUALS - WOLLASTON LAKE COMMUNITY

Exposure Pathway	Individual Doses (mSv/a)
External	
Air Immersion	0.430E-3
Ground Contamination	0.459E-3
Beach Exposure	0.120E-7
Water Immersion	0.396E-9
Inhalation	
Particulates	0.200E-2
Radon	0.253E-1
Ingestion	
Drinking Water	0.134E-4
Fish	0.153E-4
Rice	0.347E-6
Vegetables	0.442E-6
Meat	0.677E-2
TOTAL	0.350E-1

daughters are delivered to the basal cells of the bronchial epithelium. All internal doses in Table S.1 are committed effective dose equivalents computed according to the methods of ICRP 26. Recently published weighting factors for skin and lung were used in calculating total effective dose equivalents (ICRP 33).

S.3 Collective Doses

The following components of collective doses were separately considered:

1) collective doses from airborne emissions to the local population within 80 km of the three mine sites (the "Airshed Population" component);
2) collective doses to the populations along the Athabasca and Reindeer drainage basins from liquid discharges from the three mine sites (the "Watershed Population" component);

3) 'Average' lifestyle habits estimated for the population of the Wollaston Lake Community, for purposes of calculating collective doses. Not all members of this community will have the food consumption and general activity habits of the most highly exposed individuals. (See Table S.2).

Total collective effective dose equivalents are summarized in Table S.2. The collective dose from fish consumption is estimated for commercial and sport fishery catches. The total whole-body collective effective dose is 0.025 person-Sv/a including weighted dose equivalents received by skin and the bronchial epithelium. Table S.2 includes doses from both aquatic and atmospheric pathways.

TABLE S.2 COLLECTIVE EFFECTIVE DOSE EQUIVALENTS FROM URANIUM MINING IN NORTHERN SASKATCHEWAN

	Collective Doses (person-Sv/a)		
Pathways	Airshed Population	Watershed Population	Total
External			
Air Immersion	0.121E-2		0.121E-2
Ground Contamination	0.373E-3		0.373E-3
Beach Exposure		0.513E-8	0.513E-8
Water Immersion		0.586E-9	0.586E-9
Inhalation			
Particulates	0.144E-2		0.144E-2
Radon	0.187E-1		0.187E-1
Ingestion			
Drinking Water		0.530E-4	0.503E-4
Fish		0.184E-4	0.184E-4
Rice		0.139E-5	0.139E-5
Vegetables	0.295E-6		0.295E-6
Meat	0.532E-2		0.532E-2
TOTAL			0.251E-1

S.4 Doses to Biota

Radiation doses were calculated for selected terrestrial and aquatic biota. The main uncertainties in the calculations lie in the choice of (a) transfer factors to forage and from diet to organs and tissues in the case of terrestrial biota, and (b) bioaccumulation factors in the case of aquatic biota. No relative biological effectiveness weightings were used for alpha and beta radiations; accordingly, all doses to aquatic biota are absorbed dose and are expressed in grays.

Terrestrial Biota. Doses for these species were calculated for the vicinity of the Rabbit Lake site and are summarized in Table S.3. Caribou were assumed to forage in the areas around the site. Doses to moose and beaver from the ingestion of aquatic plants from Wollaston Lake were calculated. Radon inhalation exposure was not calculated since a suitable dose conversion factor for animals is not available. Doses to bone were highest due to the fact that most of the radionuclides involved, viz., Ra and Pb, are "bone-seekers". Doses to organs such as kidney and liver were lower whereas doses to muscle were generally the lowest. As a general conclusion, it may be stated that the maximum bone dose received by mammals is less than 3.5 mGy/a and that no single organ other than bone is likely to receive a dose in excess of 0.9 mGy/a. Doses to lichens were calculated to be less than 2.1 mGy/a.

TABLE S.3 DOSE RATES TO TERRESTRIAL BIOTA IN VICINITY OF RABBIT LAKE OPERATION

Exposure Pathway		Dose Rate (μGy/a)
Total		
External	- lichens	0.210E+1
	- animals	0.561E-2
Ingestion		
Caribou	- all soft tissue	0.155E+1
	- bone	0.343E+1
Moose	- all soft tissue	0.399E-1
	- bone	0.583E-1
Beaver	- all soft tissue	0.134E-1
	- bone	0.248E-1

Aquatic Biota. The tailings discharge from the Eldor Resources operation flows directly into Otter Bay before reaching the body of Wollaston Lake. Doses were calculated for aquatic biota exposed directly to the discharges. Dose rates from contaminated sediment were considered. Dose to fish was in the order of 0.02 mGy/a. They received negligible doses from contaminated sediments. Dose to fish eggs, which are generally not laid on fine sediments, was around 0.01 mGy/a, assuming an incubation period of 30 days. Calculated fish egg doses are less than one hundredth of the minimum dose at which radiation-induced abnormalities are detectable. Calculated doses for aquatic biota are shown in Table S.4.

TABLE S.4 DOSE RATES TO AQUATIC BIOTA EXPOSED TO RABBIT LAKE TAILINGS DECANT EFFLUENT

Exposure Pathway	Dose Rate (mGy/a)
Dose From Radioactivity in the Organism and in Water	
Phytoplankton (unicellular or colonial 50 μ diameter)	0.367E-0
Invertebrates (rotifers, cladocerans, copepods)	0.138E-0
Fish (pike, lake trout, whitefish)	0.154E-1
Fish Eggs (pike, lake trout, whitefish)	0.130E-1
External Dose From Contaminated Sediments to Benthic Invertebrates	
Beta	0.711E-0
Gamma	0.450E-1

S.5 Recommendations

The assessment of radiological impact carried out in this study indicates that the calculated dose to a hypothetical member of a critical group from presently operating uranium mining and milling facilities is well below maximum permissible exposure levels. Similarly, the doses to natural biota are well below levels at which effects on individuals or populations would be expected. The parameters used in much of the modelling in this study were judged to be of a conservative nature, meaning they would lead to an overestimate of dose actually received. Nevertheless, there is a degree of uncertainty associated with these calculations, and the following recommendations focus on reducing that uncertainty. This study does not include a sensitivity analysis nor an uncertainty analysis. Consequently, no firm value can be placed upon the increased level of precision associated with the adoption of any particular recommendation.

1) **Atmospheric Source Terms.** There is a need to refine the estimates of airborne release source terms by means of carefully designed field measurements and by improved modelling which takes conditions in northern Saskatchewan into account. Estimates of radon and particulate releases from tailings and open pits should make better allowance for the relatively wet conditions in summer and snow-covered or frozen grounds in winter. Better site-specific data are needed for tailings particle size distribution, radioactivity distribution, and moisture content.

2) **Aquatic Source Terms.** There is a need to verify and improve aquatic source terms by means of site-specific measurements, particularly for mines with high ore grades. In the present study, releases were estimated based upon routine monitoring data. Complete data were not available for all radionuclides. More data on releases of thorium-230, lead-210 and polonium-210 are required if better estimates of aquatic radionuclides are to be made. There is also a need to evaluate the effect of process failures or upsets on the aquatic source term. The aquatic source terms for non-conventional tailings management schemes have yet to be verified.

3) **Hydrological Data.** More detailed physical data such as dimensions of lakes and rivers, and flow rates for watersheds in the area are required for a more refined aquatic modelling.

4) **Aquatic Radionuclide Transport.** The behaviour of the uranium series radionuclides in the aquatic environment of northern Saskatchewan should be investigated. Data should be obtained on the sediment/water distribution coefficients, correlations between concentration levels in sediment and water in lakes and streams, removal rates of radionuclides by adsorption and sedimentation processes, leaching of radionuclides from sediments, sediment resuspension, and transport processes. It is recommended that site-specific bioaccumulation factors be obtained for the uranium series radionuclides in aquatic biota. Available literature data are scanty and are usually based on studies in environmental conditions substantially different from those in northern Saskatchewan. Where northern Saskatchewan data were available, these were used in this study to verify or supplement other published literature.

5) **Aquatic Modelling Detail.** The aquatic dispersion model used in this study assumes uniform mixing in the lakes and is intended to predict long-term trends in levels of radioactivity in the watersheds. Wollaston Lake, and ultimately Lake Athabasca, will likely receive drainage from most future mines. It is recommended that aquatic dispersion and radionuclide behaviour in these two lakes be modelled in more detail, taking into account local flow regimes, mixing patterns, sediment interactions, and transport.

6) **Atmospheric Radionuclide Transport.** It is recommended that more site-specific environmental transfer data be obtained for the uranium series radionuclides in airborne releases to terrestrial vegetation, particularly animal forage. Better transfer data for animal diet to animal tissue should also be obtained for the more important species in northern Saskatchewan such as caribou, moose, beaver, muskrat, and waterfowl.

7) **Animal Dosimetry.** To improve the assessment of doses to biota, better animal dosimetry models and data are required, particularly in the case of terrestrial animals.

8) **Meteorology Data.** More site-specific meteorological data are required to improve the accuracy of atmospheric dispersion calculations. In particular, data are needed on the joint frequencies of atmospheric stability, wind direction and speed, as well as atmospheric mixing heights.

9) **Area-specific Diffusion Data.** Available Stability Category classification and associated dispersion coefficients are usually based on studies in environmental conditions substantially different from that in northern Saskatchewan. Hence it is recommended that diffusion data typical of the area be obtained.

10) **Specific Data on Area Inhabitants.** To improve the assessment of doses to humans, more site-specific habit survey data on the local populations are required.

11) **Post-abandonment Radiological Impact.** Modelling studies have not been carried through to the post-abandonment phase of tailings disposal. The various tailings management options should be examined to determine the effect they might have on the regional aquatic and terrestrial environments in the long term. Radiological impact in the post-abandonment phase of tailings disposal could possibly be more significant than in the operational phase.

1 INTRODUCTION

1.1 Objectives and Scope

Objective 1: To estimate the radiological impact of uranium mining in northern Saskatchewan.

This study provides a first approximation of the integrated radiological effects of three uranium mine and mill complexes operating simultaneously in northern Saskatchewan from the early 1980's to the year 2000. The study also provides the methodology for the incremental addition of any number of new operations and calculations of the radiological dose from these operations when site-specific data become available.

The large commercial deposits of uranium ore in northern Saskatchewan will prompt the development of mines and mills for the extraction and processing of this natural resource. With this new development, as well as the existing facilities, interest has turned to defining the potential impacts these developments will have on the region. Generally speaking, the environmental impacts can be divided into two major categories, radiological and non-radiological, referring to impacts caused by the release of radioactive materials to the environment and the discharge of other non-radioactive substances, respectively.

Objective 2: To develop a general methodology for assessing radiological impacts of uranium mining.

This study has developed a methodology for assessing the radiological impacts of the uranium industry in northern Saskatchewan. The work was limited to consideration of the three existing operating mines. The methods and analysis can be extended to other ore body developments, but due to the uncertainty of many of the project parameters for these developments, this was not done. The methodology provides an assessment of the regional impact of the developments and provides some appreciation of the magnitude of the radiological doses to individuals and biota in the area. Only the uranium series radionuclides are considered in this study.

Before undertaking any assessment, it is necessary to identify the source of releases into the biosphere in terms of location, quantity, and receptor media. Following this step, the emissions are modelled as they progress through the receiving media. Both air dispersion modelling and modelling in the aquatic environment are discussed in this report. The modelling provides estimates of concentrations at various locations and enables doses to individuals and biota to be determined.

1.2 General Terms of Referrence

This study was intended to provide a regional overview based upon the best available information at the time of writing. Use of any of the modelling information generated in this study for estimating radiological impacts in the immediate proximity of the facilities should be discouraged. The present models cannot be applied in this context.

The population data used in this study is for 1981, the latest census data available. All emissions data are based on expected releases from mining and milling when each facility has reached mature operation, anticipated for the mid-1980's. Abandoned tailings and waste rock areas were not considered as source terms for regional modelling.

1.3 General Background

Despite the recent reduction in the growth rate of nuclear power, uranium mining is expected to expand in northern Saskatchewan before the year 2000 due to the abundance and high grades of uranium ore bodies in the region which make their exploitation economically attractive. The development of Eldorado's Collins Bay "B" Zone deposit at their Rabbit Lake facility points to the future expansion of mining. Development of further ore bodies is also expected to continue as present mines are worked out. Radioactive releases from future operations, however, are not included in this study.

Combined Radiological Releases. Associated with the operational and projected mining developments are sources of radionuclides and other potentially toxic process- and ore-derived materials. Many of the uranium deposits in northern Saskatchewan are unique in that the grade of ore found in many parts of the deposits and, in some cases, mined and milled directly is very high (Table 1). The relatively high grade of ore and large number of deposits within northern Saskatchewan give the area a large percentage of total world reserves.

Each company wishing to begin operations in northern Saskatchewan is required to submit an Environmental Impact Assessment to the Government of Saskatchewan Department of the Environment in which the effects of released radionuclides and other toxic chemicals on the natural and human environment are

TABLE 1 SUMMARY OF OPERATING MINES IN NORTHERN SASKATCHEWAN

Operator	Location of Mine and Type	Estimated Reserves Tonnes U* (as U_3O_8)	Milling Operation and Type	Average Ore Grade** (% U_3O_8)	Operational Life
Eldorado Resources	Collins Bay (open pit)	20 000	Acid Leach	0.03-36.0 (0.45)	8 years from 1982
Amok Phase II	Cluff Lake (open pit)	10 000	Acid Leach	0.04-0.6 (0.36)	11 years
Key Lake Mining Corporation	Key Lake (open pit)	60 000	Acid Leach	0.1-10.0 (2.5)	15 years

* The estimated reserves are very approximate figures and subject to revision as new data become available.

** The average ore grade appears to vary for most of the mines from a lower cut-off of less than 0.1% to very high local concentrations which may be as high as 50% but are usually blended with lower grade ore before processing (ore grade % U_3O_8) = value used for modelling.

predicted. However, each of these operations deals only with the impact from its particular operation. This study attempts to assess the combined radiological effects of the three mine and mill complexes that will continue operation in northern Saskatchewan during the next decade.

Data Sources. The data used in this study were derived from information supplied by the mining companies, the provincial and federal governments (Saskatchewan Department of the Environment, Saskatchewan Department of Energy and Mines, AECB, and Environment Canada), and published literature. Where actual monitoring data were made available, they were used in the calculations. In other instances, where no monitoring data were available, it was necessary to scale the output terms to existing operations or use data generated in the United States and scale them to northern Saskatchewan conditions. In some cases, when deciding upon such terms as transfer coefficients or bioaccumulation factors, little or no directly relevant data were available. In these cases, conservative values were assumed, based upon literature values and adjusted to suit the northern Saskatchewan environment. The purpose of this approach was to provide as realistic an assessment as possible of the radiation dose to aquatic biota, terrestrial biota, and human populations in the area. In some instances, sensitivity analyses were carried out in order that the effect of some of the estimated parameters could be evaluated.

The mines are located, for the most part, in relatively uninhabited areas. A study area map is provided as Figure 1. At present, there are no plans for two sites to be located near any of the known mining operations.

The atmospheric emissions from each facility were modelled in detail for the area within an 80 km radius of each operation (Figure 1). These areas encompass a few permanent population centers populated mainly by native peoples. Some seasonal population centers, either fishing lodges or supply depots, were identified within the same 80-km radii.

It is recognized that the gaussian plume diffusion model used in this study has limitations when used to predict radionuclide air concentrations more than about 15 km from a source. The use of more complex, sophisticated dispersion models is also fraught with uncertainty. The use of such models was considered beyond the scope of this study since their own uncertainties render them no more reliable at extreme ranges than the gaussian model used. The rough estimates, out to an 80 km radius for each site, are considered adequate for the present purpose, in view of the uncertainties involved in assessing radiological impact.

1.4 Report Structure

The basis of the methodology utilized in developing this assessment is complex. The quantities of radionuclides released to both atmospheric and aquatic pathways were determined. Their dispersion and dilution in the appropriate medium were modelled as were the uptake rates of the various elements. Once environmental levels in all media were available, it was possible to calculate the radiological impact of the aquatic and atmospheric releases. Finally an assessment of the doses to terrestrial and aquatic biota was completed.

The derivation of source terms and the dispersion of the radionuclides in the environment are presented in detail in Appendixes A and B. The impacts of the releases are derived in Appendixes C and D. The details in the appendixes have been summarized in the report that follows. The emissions are discussed in Chapter 2. Chapter 3 presents the environmental setting for the study. The theory of dose calculations is dealt with in Chapter 4 while Chapters 5-7 present the results of the calculations.

The approach provides some overlap between the text and the appendixes. This is particularily apparent in the tabular data. This overlap has allowed the authors to simplify the tables in the text. The reader is referred to the appendixes for details of all assumptions and references used to derive the data presented.

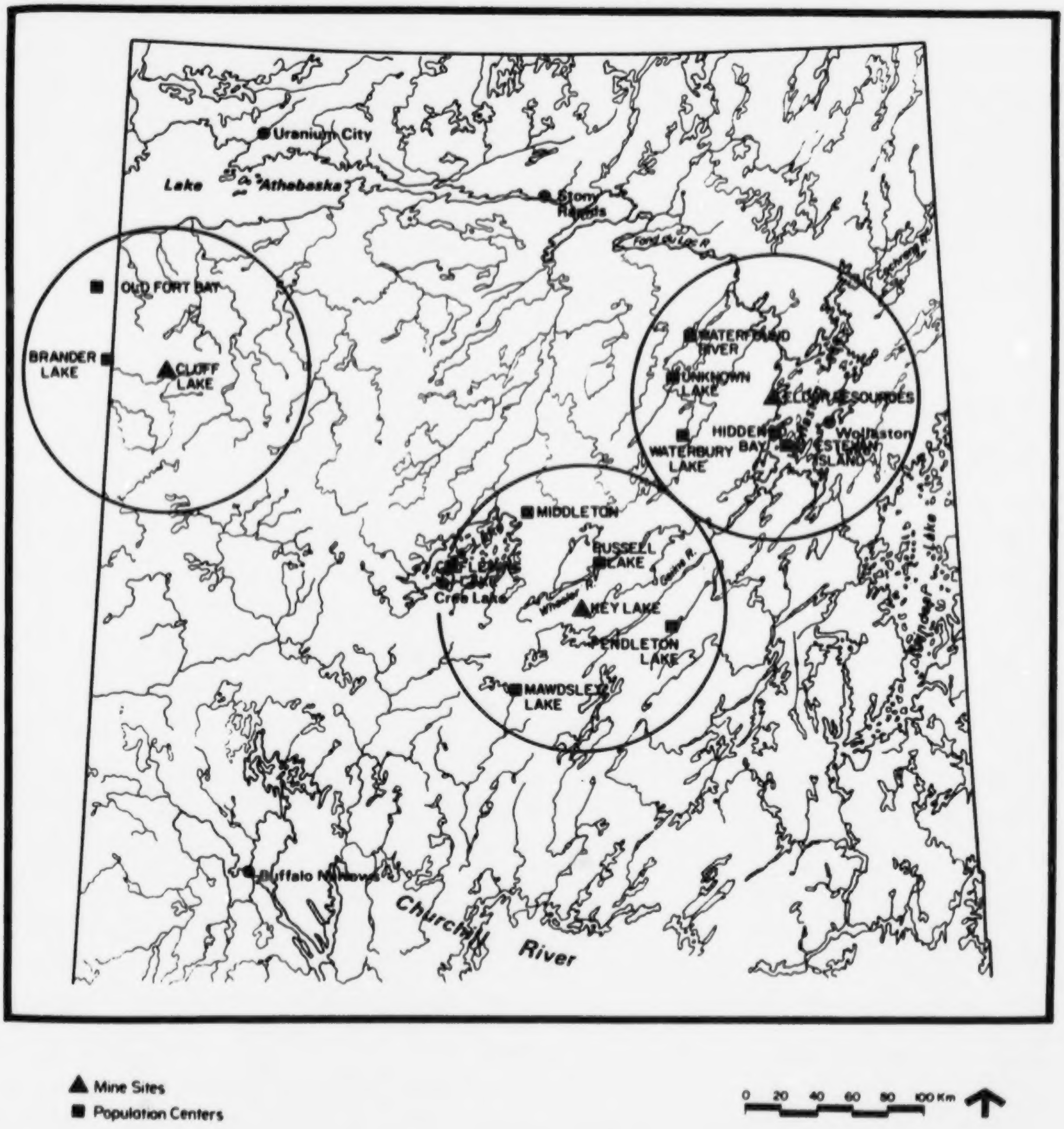
Uranium City
Lake Athabaska
Stony Rapids
Fond du Lac R.
Cochrane R.
OLD FORT BAY
BRANDER LAKE
CLUFF LAKE
WATERFOUND RIVER
UNKNOWN LAKE
ELDOR RESOURCES
HIDDEN BAY
Wollaston
WESTERN ISLAND
WATERBURY LAKE
MIDDLETON
RUSSELL LAKE
Geikie R.
Wheeler R.
Cree Lake
KEY LAKE
PENDLETON LAKE
MAWDSLEY LAKE
Buffalo Narrows
Churchill River
Mine Sites
Population Centers
0 20 40 60 80 100 Km

FIGURE 1 CRITICAL RECEPTOR LOCATIONS IN THE STUDY AREA

2 SOURCES TERMS

2.1 Derivation and Use of Sources Terms

Derivation. In this assessment of the radiological impact of uranium mining and milling operations in northern Saskatchewan, many of the source terms must be estimated because long-term operational data are not available. Since all the analysis is built on a definition of these source terms, care had to be taken in arriving at these numbers. Where long-term monitoring data from operational facilities were available, source term estimates are the most accurate; however, for facilities which have a short operational history or are just starting operation, the source term estimates are less precise.

Questionnaires requesting source term information were sent to the presently operating mines. The most detailed information for the operating or new mines is available from the data in the impact assessments filed by the mining companies. These assessments provided the basis for estimates for Amok at Cluff Lake, Key Lake Mining Corporation at Key Lake, and Eldor Mines at Rabbit Lake (Collins "B"). Even with these data, some assumptions had to be made regarding unspecified source terms. Any future application of the methodology presented in this report should be prefaced by an updating of the source terms to reflect the actual operational conditions and design available at that time.

The source terms are presented for mining, milling, and tailings management operations, including both atmospheric and aqueous releases. Aqueous releases are presented for mining and tailings water management. The aqueous release from the milling operation generally goes to the tailings area and is therefore considered part of the tailings discharge fluid.

Use of Source Terms. The atmospheric source terms generated as part of this study were input to both the UDAD and MILDOS codes. The aquatic source terms were developed for input to an aquatic model developed as part of this study.

2.2 Aqueous Releases

2.2.1 Mines. The three mines considered in this study all have or are planned to have open pit operations from which the ore is mined. Depending upon the amount of seepage water projected to enter the open pit, there may or may not be a ring of interceptor wells or trenches surrounding the pit. Some sites will also have piping to divert creeks from the pit area in order to reduce infiltration potential. The purpose of these cutoffs is to intercept water flowing in from the country rock surrounding the ore body before it becomes contaminated through contact with the ore and pit workings. Data collected at Eldor's Rabbit Lake operation (Zone B), at Amok's Cluff Lake operations, and at Key Lake Mining Corporation's Key Lake project (Table 2) suggest that although interceptor flow may be significant, the water quality associated with these projected flows does not differ significantly from background levels in the area. Accordingly these flows are now being or are predicted to be discharged directly to the receiving environment without treatment.

This interceptor water rises principally from flow to a local groundwater depression caused by the creation of an open pit and is comprised of both water that is drawn from storage in the surrounding overburden and bedrock and water which infiltrates from nearby streams, ponds and lakes. This, accordingly, is the water which would naturally discharge to local surface flow systems and, as such, any radionuclide loading associated with it should be considered as part of the natural background. The mass loading arising from untreated interceptor water is accordingly not considered as part of the incremental source loading from the mine and mill complex.

A certain fraction of the infiltrating water and direct precipitation onto the open pit area will not be picked up by interceptor wells or trenches and will become contaminated through contact with the exposed ore. It is collected as pit sump water and must be treated before release to the open environment. This water in all instances is either treated or assumed to be treated within the tailings treatment system. Before leaving the tailings treatment system, if will accordingly take on the chemical characteristics of treated tailings decant water. The loadings from this source are in contrast to those from the interceptor water considered as an incremental loading of radioisotopes to the environment and a direct result of the mining and milling operation.

Depending upon the availability of pit sump water, it may be used as makeup water for the mill circuit. If it is not plentiful enough or if it is not economical, fresh makeup water may be used from a nearby surface water source or well. This makeup mill process water will, after passing through the mill circuit, be treated as part of the tailings decant treatment system.

All other waters that may become contaminated on site, such as runoff from ore storage stockpiles, are to be collected and treated as part of the tailings treatment system if the trace element concentration is above background.

TABLE 2 RADIONUCLIDE CONCENTRATION IN WATER SAMPLES FROM THE THREE MINING SITES

Water Quality Trench Water Parameter	Cluff Lake				Key Lake			Rabbit Lake	
	Local Surface Water		Interceptor Well Water		Local Surface Water		Interceptor Well Water	Local Surface Well Water	Interceptor Well Water
	Range	Avg.	Range	Avg.	Range	Avg.	Avg.	Range	Range
Uranium total (μg/L)	0.1-1.0	0.42	0.5-31	4.8	0.5-1.0	0.5	50	0.06-1.60	0.8-2.65
Thorium-230 (Bq/L)	0.02-0.04	0.02	0.01-0.04	0.02	0.04-0.18	0.03	0.02	0.004	0.004-0.05
Radium-226 (Bq/L)	0.004-0.03	0.005	0-1.1	0.15	0.004-0.18	0.02	0.07	0.004	0.02-0.09
Polonium-210 (Bq/L)	0.004-0.04	0.01	0.004-.13	0.03	0.004-0.18	0.02	0.07	0.004-0.015	0.004-0.01
Lead-210 (Bq/L)	0.004-0.07	0.02	0.02-0.59	0.08	0.004-0.02	0.04	0.3		0.002-0.16
pH	7-8				5.1-7.3	6.5	6-7	5.9-7.1	5.9-6.5
Total Dissolved solids (ppm)	33-136	75	(62-208)	117	15-75	25		10-45 (70 outlier)	
Discharge Rate (L/s)							500		50

Ranges and averages for Cluff Lake interceptor wells are based on samples collected at the north, south and west wells from January 1980 -September 1982 (AMOK 1976 and 1982). Other values are based on the environmental impact assessments associated with each site (KLMC 1979).

Other aqueous wastes such as sanitary wastes will be treated separately as they are not expected to contain radionuclide concentrations above background and are not considered to be contributing to any significant increase in radioactivity loading.

2.2.2 Tailings Discharge Fluids. The aqueous radioactivity loadings from the three mines being evaluated in northern Saskatchewan arise from the tailings decant treatment system.

The loadings from Eldor's Rabbit Lake operation are based upon average flow volumes and discharge concentrations as monitored from January 6, 1981 to October 14, 1982. The loadings from Amok's Cluff Lake operation are based upon average flows and radionuclide concentrations are monitored from June 1, 1982 to October 20, 1982. These data represent the most recent measurements available at the time of this study. The last few measurements in the Cluff Lake series suggest a possible increasing trend in uranium loading. The highest concentration recorded in October exceeds the mean by a factor of three. The possible consequences of underestimating uranium loadings are discussed in Section 6.4.

The loadings assumed for the Key Lake Mining Corporation's Key Lake facility are based upon the assumption that discharges will have to meet Saskatchewan draft regulations. In the absence of applicable Saskatchewan draft regulations, federal regulations were assumed to apply. In the absence of operational data, it is assumed, for the purposes of this study, that discharge concentrations will be below the appropriate regulatory maximum and above values from other mines from which operational data are available. Discharge volumes for the Key Lake facility are estimated from data presented in the Environmental Impact Assessment.

Discharge volumes and concentrations of radionuclides upon which loadings are based for these three facilities are presented in Table 3. This table shows a discharge concentration, a background concentration, and a net incremental concentration. It is the net incremental concentration that is considered to be the incremental concentration loading from the mine mill facility. This value, when coupled with the average discharge volume, results in the incremental loading value.

2.3 Atmospheric Emissions

This section presents the quantities of radioactive material released to the atmospheric environment as a result of mine and mill operations. All phases of the mining operation are considered. Calculated quantities of radioactive material released are called "source terms". These source terms include gaseous and particulate emissions from

TABLE 3 DISCHARGE VOLUMES AND CONCENTRATION OF RADIONUCLIDES IN TAILINGS DECANT WATER DISCHARGE

	Mine and Mill Facility									
	Cluff Lake*			Key Lake			Rabitt Lake*			
Water Quality Parameter	Discharge	Background	Net	Discharge	Background	Net	Discharge	Background	Net	Prov. Std.
Uranium total (µg/L)	341.4	.42	340	1000.**	1	1000	1021.6	1	1000	1000
Thorium-230 (Bq/L)	0.02	0.01	0.01	0.18	0.02	0.16	0.04	0.004	0.04	3.7
Radium-226 (Bq/L)	0.056	0.005	0.05	0.37**	0.02	0.35	0.3	0.004	0.3	0.37
Polonium-210 (Bq/L)	0.07	0.02	0.05			0.2	0.025	0.004	0.02	no draft prov. regs.
Lead-210 (Bq/L)	0.04	0.02	0.02	0.2	0.04	0.16	0.095	0.004	0.09	1.85
Discharge Rate (L/s)	45			45.3			80			

* The measured values are average concentrations based upon a composite of company and Saskatchewan Department of Environment samples and measurements. Cluff Lake averages are based on 13-57 final effluent samples taken from June-October 1982. Rabbit Lake averages are based on 11-107 samples of precipitation pond discharge taken from January 1981 to October 1982. Rabbit Lake flow figures are based upon total treated flow from present mine and mill (i.e., no interceptor).

** Until actual operational data are available it is assumed that discharge concentrations are at regulated maxima. Other values for Key Lake are based upon the upper limits observed at other facilities.

a variety of sources such as mill vents, haul roads, open pits, waste rock piles and tailings areas.

Source terms cannot be directly and completely measured. They must be calculated from sampling data and estimates. Appendix A, Section A.2 provides details of the calculation of atmospheric emission source terms presented in Tables 4 to 9.

TABLE 4 RADON EMISSIONS - KEY LAKE*

Item		Exposed Area (m^2)**	Concentration U_3O_8(%)	Rn Release (TBq/a)
Ore Storage	- Regular	6.5×10^3	2.5	76
	- Cobble	1.0×10^5	0.7	230
Waste Storage (Gaertner)	- Special	1.2×10^4	0.04	2
	- Rock	4.0×10^5	0.015	28
	- Sand	1.7×10^5	1.4×10^{-5}	-
Mining (Gaertner)	- Rock	6.0×10^4	0.015	4
	- Ore	2.0×10^4	2.5	230
	- Overburden	7.2×10^4	1.4×10^{-5}	-
Mining (Dielmann)	- Rock	3.0×10^5	0.015	21
	- Sand	9.9×10^4	1.4×10^{-5}	-
Waste (Dielmann)	- Rock	Negligible	0.015	-
	- Sand	7.9×10^5	1.4×10^{-5}	-
Dewatering	- Gaertner	$0.5\ m^3/s$	185 Bq/L	3
	- Dielmann	$0.5\ m^3/s$	185 Bq/l	3
Material Handling	- Ore	3 @ $1.2 \times 10^5\ m^3/a$	1350	
	- Waste	$4.2 \times 10^6 m^3/a$		3
	- Special waste	$1.8 \times 10^4 m^3/a$		-
TOTAL				600

* Source terms estimated at the end of mining expansion G IV (1988).

** Total area exposed for radon release.

Specific data on emissions to the atmosphere are not as readily available as are aquatic release data. The Key Lake mine information was based on detailed work presented in the October 1979 Environmental Impact Statement. Cluff Lake data were developed from both the October 1976 Environmental Assessment document (AMOK 1976) and the preliminary Phase II report (Amok 1982). The Rabbit Lake and Collins Bay

TABLE 5 DUST RELEASES - KEY LAKE

HANDLING		Production (t/a)	Capacity (m^3)	Silt (%)	Moisture (%)	Emissions (kg/a)	Concentration (% U_3O_8)
Waste	- Rock	1.5×10^6	42	1	10	1.5	1.4×10^{-5}
	- Sand	4.2×10^6	42	5	3	230.0	1.4×10^{-5}
	- Primary	1.8×10^5	17	5	10	13.0	0.015
	- Cobble	9.2×10^4	17	5	3	4.0	0.015
Ore	- Cobble	3.0×10^4	17	5	3	0.4	0.7
	- Primary	9.5×10^4	17	1	10	0.2	2.5
Reclaim	- Cobble	3.0×10^4	4	5	3	20.4	0.7
	- Primary	9.5×10^4	4	1	10	1.2	2.5
Total Handling Emissions						270.7	

ACTIVE STORAGE	Volume (t/a)	Duration (days)	Silt (%)	Emissions (kg/t)	Emissions (kg/a)	Concentration (% U_3O_8)
Primary Ore	9.5×10^4	83	5	0.0861	8200	2.5

OPEN SOURCES	Area (m^2)	Silt (%)	Erodibility (t/km^2a)	Emissions (kg/a)	Concentration (% U_3O_8)
Waste - Gaertner - rock	4×10^5	3	0.368	4.7×10^4	0.015
- sand	2×10^5	3	-	2.0×10^4	1.4×10^{-5}
- Deilmann - rock	3×10^5	3	-	3.5×10^4	0.015
- sand	1×10^5	3	-	1.2×10^4	1.4×10^{-5}
Special	1×10^4	3	-	0.1×10^4	0.04
Cobble Ore	1×10^5	5	-	1.2×10^4	0.7
Total Wind Erosion Emissions (Key Lake) (includes primary ore)				1.4×10^5kg/a	

TABLE 6 RADON EMISSIONS - (ELDOR) RABBIT LAKE COLLINS BAY DEPOSIT IN PRODUCTION*.

Source	Area (m^2)	Concentration (% U_3O_8)	Radon (TBq/a)
Collins Bay			
Mining Area			
- Ore	3.9×10^4	0.453	80
- Waste	2.4×10^4	0.05	6
- High Ore	2.0×10^3	10.0	90
- Overburden	7.8×10^4	0.001	0.4
Storage			
- Waste Rock	2.9×10^5	0.006	8
- Low Grade Ore	3.4×10^4	0.05	8
- Ore - average	2.6×10^4	0.453	50
- high	439 each @ 1.0, 3.0, 5.0 7.0, 10.0, 15.0		80
Operations			
- Trucking - ore			7
- waste			0.07
- Dewatering 2050 L/min @ 5000 pCi/L			0.2
Subtotal			330
Less allowance for area covered			2
TOTAL			328
Rabbit Lake			
Pit Area	2.3×10^5	0.001	1
Storage Area - waste	5.6×10^5	0.006	15
- low grade ore	1.3×10^5	0.05	30
- B zone ore	8.6×10^3	0.45	320
- Rabbit ore	6.2×10^4	0.36	107
Operations			
- Trucking			3
- Dewatering			0.2
Subtotal			176
Less allowance for area covered			2
TOTAL			174

Note: Haulage from Collins to Rabbit Lake estimated @ 4×10^{-3} - 50×10^{-3} TBq/a depending on loss from vehicles

* Year of reference 1987.

TABLE 7 DUST RELEASES - ELDOR MINES (1987)

COLLINS BAY

Handling	Production (t/a)	Concentration (% U_3O_8)	Capacity (m^3)	Silt (%)	Moisture (%)	Emissions (kg/a)
Waste - Primary	1.8×10^6	0.006	17	1	10	6
- Special	2.7×10^5	0.05	17	5	10	4
Ore - Trucks	4.2×10^5	0.45	17	1	10	1
- Reclaim	3.5×10^5	0.45	4	1	10	5
Total Handling Emissions						16

Active Storage	Concentration (% U_3O_8)	Volume (t/a)	Duration (days)	Silt (%)	Emissions (kg/t)	Emissions (kg/a)
Ore - High grade	1.0	1.3×10^4	5	5	5×10^{-3}	65
	3.0	1.2×10^4	5	5	5×10^{-3}	60
	5.0	5.1×10^3	5	5	5×10^{-3}	26
	7.0	5.1×10^3	5	5	5×10^{-3}	26
	10.0	5.1×10^3	5	5	5×10^{-3}	26
	15.0	2.6×10^3	5	5	5×10^{-3}	13
- Average grade	0.45	3.5×10^5	156	5	1.6×10^{-1}	5.6×10^4

Open Sources	Concentration (% U_3O_8)	Area (m^2)	Erodibility (t/km^2a)		Silt (%)	Emissions (kg/a)
Special Waste	0.05	3.4×10^4	0.368		3	3.9×10^3
Waste Rock	0.006	2.9×10^5	0.368		3	3.3×10^4
Total Wind Erosion Emissions (Collins Bay)						9.3×10^4

RABBIT LAKE

Active Storage	Concentration (% U_3O_8)	Volume (t/a)	Duration (days)	Silt (%)	Emissions (kg/t)	Emissions (kg/a)
Ore Piles	0.45	3.5×10^5	120	5	0.12	4.2×10^4

Open Sources	Concentration (% U_3O_8)	Area (m^2)	Erodibility (t/km^2a)	Silt (%)	Emissions (kg/a)
Rabbit Ore	0.36	6.2×10^4	0.368	5	1.2×10^4
Waste Rock	0.006	5.6×10^5	0.368	3	6.4×10^4
Special Waste	0.05	1.3×10^5	0.368	3	1.5×10^4
Total Wind Erosion Emissions (Rabbit Lake)					1.3×10^5

Haulage Losses Collins Bay to Rabbit Lake = 340 - 3700 t/a ore or 4×10^{-3} - 50×10^{-3} TBq/a U238

TABLE 8 RADON EMISSIONS - AMOK (mid-1987)

Source	Area (m^2)	Grade (% U_3O_8)	Radon (TBq/a)
Abandoned D Pit	1.2×10^4	0.0002	7.4×10^{-3}
Claude Pit			
Ore	2.3×10^4	0.36	40
Sub-ore	2.3×10^4		2
Overburden	4.5×10^4	0.0002	0.04
Storage (Claude)			
Overburden	1.2×10^5	0.0002	0.1
Sub-ore	1.1×10^5	0.015	9
Ore	3.2×10^3	0.28	5
Underground			
Waste	5.4×10^3	0.015	0.4
Ore	3.2×10^3	0.28	5
Ventilation			0.5
Dewatering (Claude)			0.1
Vehicular Releases			25
Subtotal			87
Less allowance for existing cover			0.2
Total			87

Phase B developments were detailed in the Phase B impact statement (Gulf 1980). Rabbit Lake, including the Collins Bay "B" zne deposit, is now owned by Eldorado Resource.

In some cases, the lack of specific data required that a common approach be developed for all sites. Of particular use in developing this approach were: the Final Generic EIS for Uranium Milling (NUREG-0706 1980) and the Battelle publication, Prediction of the Net Radon Emission from a Model Open Pit Uranium Mine (NUREG/CR-0628 1979).

The terms were developed for three broadly split activities which define the operations. In this way, similarities could be identified to aid in extrapolating values for each site from known emission values. The broad categories dealth with were:

TABLE 9 DUST RELEASES - AMOK (mid-1987)

HANDLING

Source	Production (t/a)	Concentration % U_3O_8	Capacity (m^3)	Silt (%)	Moisture (%)	Emissions (kg/a)
Ore	2 @ 1.1 x 10^6	0.36	17	5	10	103
Waste	1 @ 2.3 x 10^6	0.015	17	10	10	76
Reclaim	1 @ 1.1 x 10^6	0.36	2	5	8	90
Total Handling Emissions						269

ACTIVE STORAGE

Source	Concentration % U_3O_8	Volume (t/a)	Duration (days)	Silt (%)	Emissions (kg/t)	Emissions (kg/a)
Ore	0.36	1.1 x 10^6	30	5	3.2 x 10^{-2}	3.5 x 10^4

OPEN SOURCES

Source	Concentration % U_3O_8	Area (m^2)	Erodibility ($t/km^2/a$)	Silt (%)	Emissions (kg/a)
Waste	0.015	1.1 x 10^5	0.368	5	2.2 x 10^4
Overburden	2 x 10^{-4}	1.2 x 10^5	0.368	10	4.9 x 10^4
Total Wind Erosion Emissions					1.1 x 10^5
Mine Ventilation Area = 5.8m^2	Flow = 47 m^3/s	Release U-238	0.174 Bq/s		

(a) mining
(b) milling, and
(c) tailings management

As stated in the introduction, there are numerous ways to arrive at source emission estimates. In order of descending accuracy these include:

- direct measurement
- mass balance calculations based on process and control device information, and
- emission factor data

Some inherent problems exist with each approach:

- it is difficult and costly to accurately define the source strength for wind erosion and fugitive emissions;
- mass balance and control device information are not developed uniformly for all mines, and
- emission factor data is subject to site-specific irregularities not always identified in the source documentation.

The source terms developed in this report recognize some of these issues. They are developed on the basis of a uniform approach at each site. Where site-specific data are available, comparisons are made to other approaches in order to fairly present the data from each location.

Mining Operations. Two types of mining operations are used for uranium extraction in northern Saskatchewan. Underground mining will be used at Cluff Lake; open pit mining methods are used at all three sites. Both approaches cause increased emissions of radon gas and radioactive particulate matter with respect to natural levels. Operations involved in ore removal create dust and increase radon releases. For open pit mines these activities are sheltered from direct wind flows and an area of recirculation may be created in the pit.

Milling Operations. Milling operations are similar at all three locations:

- crushing
- grinding
- leaching
- yellowcake packaging

It is thus possible to compare the emissions from the above mill sources for each site on the basis of ore grade.

Limited literature data is available to substantiate any emission factor chosen for these sources. A summary of mill emissions used for modelling is presented in Table 10.

Tailings Management. The MILDOS code provides a method of determining the tailings pile source strength for particulate emissions. Calculated radon source strengths are input into the programme. Source data for the code were compiled to reflect northern Canadian climatic conditions.

Wind erosion source terms for the milling operations were developed on the basis of experimentally derived equations for a storage pile. For tailings areas, the MILDOS code uses classical fluid mechanics principles combined with the work of Bagnold, Belly, Lettan, Gillette, and Travis to predict wind erosion source terms (NUREG-2011 1981). See Table 11 for a summary of tailings area emissions.

TABLE 10 MILL EMISSIONS

Mill		KLMC		ELDOR		AMOK	
Production Rate							
Ore (t/d)		720		1600		1500	
Yellowcake (t/d)		18		7.2		5.4	
Emissions Source	Emission Factor	Particulate Matter (kg/d)	Radon (TBq/a)	Particulate Matter (kg/d)	Radon (TBq/a)	Particulate Matter (kg/d)	Radon (TBq/a)
Crusher	0.03	21.6	3	48.0	1	45.0	0.3
Grinder	0.02	14.4	0.2	32.0	0.1	30.0	0.04
Leach	2.1×10^{-3}	1.5	2.0	3.4	0.8	3.2	0.2
Yellowcake	0.09	1.6	-	0.7	-	0.5	-

TABLE 11 TAILINGS MANAGEMENT MODELLING PARAMETERS

Parameter	Key Lake	Cluff Lake Phase II	Eldor Mines
Ore activity (BqRa/g)	260	37	47
Area for tailings disposal (ha)	50	94	50
Particle density (g/cm^3)	2.4	2.4*	2.4*
Median diameter (cm)	0.006	0.03*	0.01
Ht of wind monitor (cm)	1 000	1 000	1 000
Surface roughness ht (cm)	0.1	1.0*	1.0*
P, particle mass fraction 20 μm (%)	15	20	30
Wind erosion scale factor (ratio of whole)	0.33	0.2	0.4
Bulk Specific Activity			
^{238}U (Bq/g)	18	26	3
^{230}Th (Bq/g)	246	36	45
^{226}Ra (Bq/g)	259	37	47
^{210}Pb (Bq/g)	171	37	47
Activity Fraction 20 μm	0.86	0.4*	0.4*
W, moisture content %	30	30	40
Effective radon source area (ha)	50	19	20
Radon emission summer (TBq)	190**	70	0.1×10^{-3}
Radon emission winter (TBq)	19**	7	0.1×10^{-4}
Total (TBq/a)	171	63	0.1×10^{-3}

* MILDOS default value.

** Scaled on basis of total value provided by KLMC (1979).

3 NATURAL ENVIRONMENT OF NORTHERN SASKATCHEWAN

The dispersal and migration of radioisotopes through the environment are functions of the conditions within each compartment of the geosphere, hydrosphere, atmosphere, and biosphere. In northern Saskatchewan, particular attention must be paid to the seasonal conditions within the natural environment. During the winter period extending from October to May, snow cover reduces natural aerial source terms. Freeze-up reduces dilution potential; cold temperatures alter the seasonal prevalence of some species, shift the diet of other species, and reduce metabolic rates. The climatic factors such as temperature, precipitation, extent of permafrost, and prevailing winds all influence the release and migration of radionuclides through atmospheric systems. The hydrological factors such as stream flow rates, trophic levels, dilution factors, residence times, water, and sediment chemistry are important in determining the radionuclide removal and uptake potentials from aqueous systems. The ecological factors such as species prevalence, niche preference, dietary and lifespan characteristics affect the potential for uptake and bioaccumulation of radioisotopes in their environment. Accordingly, these variables are briefly described for northern Saskatchewan to enable the reader to have an appreciation of the regional environmental systems. Only selected locations in the study area were examined. These were communities with permanent populations or camps with known transient populations. Table D.7 lists the communities considered for the aquatic studies. The receptor areas are shown in Table C.1. All areas and watershed boundaries are shown in Figure 2.

3.1 Climatology and Meteorology

Northern Saskatchewan is situated in the Boreal climatic zone, characterized by appreciable snow cover lasting for more than half the year and by temperature extremes common in winter and summer (Boughner and Thomas 1973). Short cold summers are dominated by cold, moist westerly air masses from the northern Pacific. Long, extremely cold winters are dominated by cold arctic air masses (Langley 1972; Hare and Thomas 1974).

The climate of northern Saskatchewan is not widely documented, primarily due to the scarcity of climatological observing stations and lack of historical weather data. Most stations have less than 20 years of climate data.

The temperature regime is characterized by an extremely cold January minimum, followed by a rapid rise from February through June, a July maximum, and a

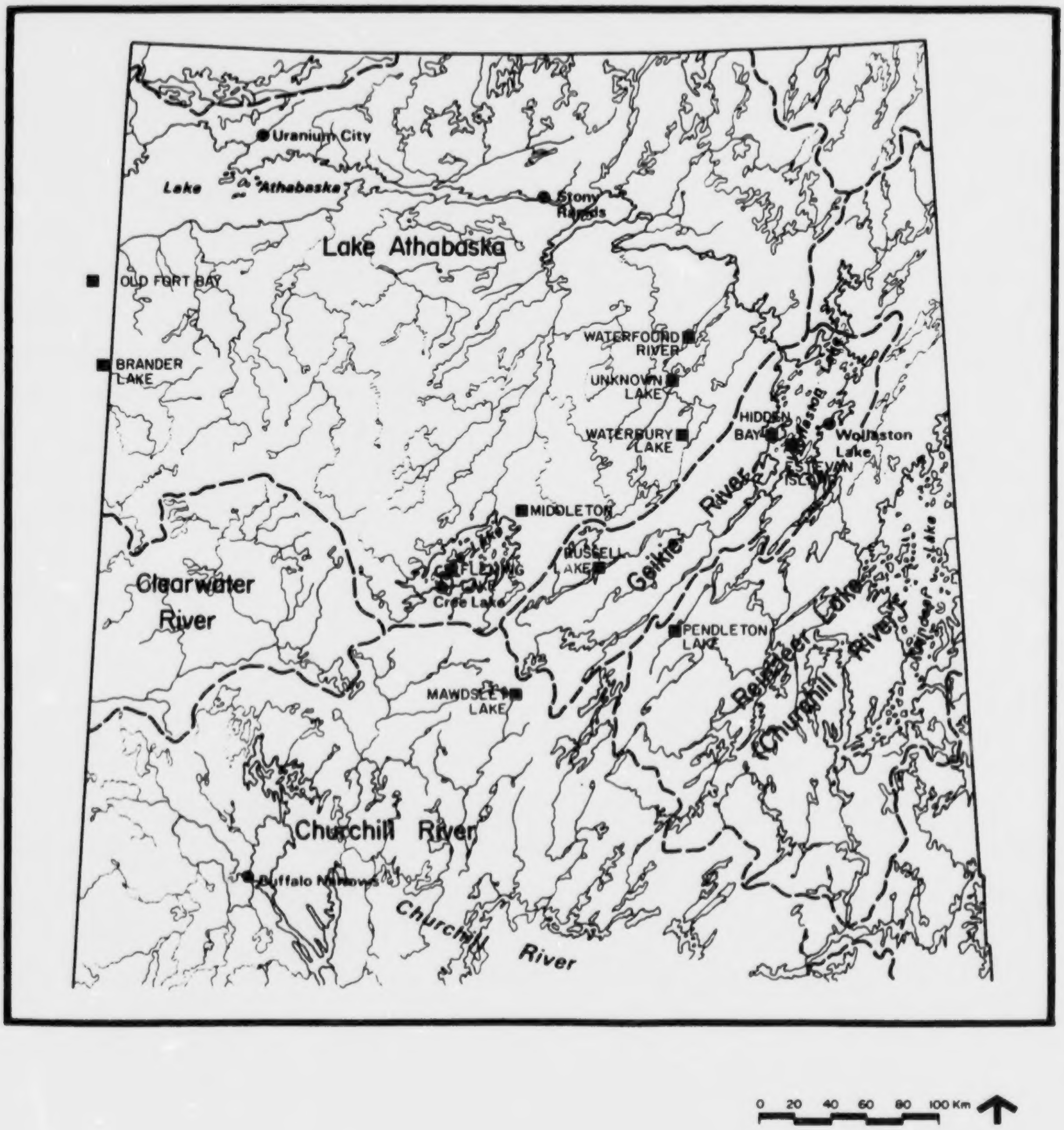
Uranium City
Lake Athabaska
Stony Rapids
Lake Athabaska
OLD FORT BAY
BRANDER LAKE
WATERFOUND RIVER
UNKNOWN LAKE
WATERBURY LAKE
HIDDEN BAY
Wollaston Lake
Wollaston Lake
ESTEVAN ISLAND
MIDDLETON
RUSSELL LAKE
Geikie River
Cree Lake
FLEMING LAKE
Cree Lake
Clearwater River
PENDLETON LAKE
Reindeer Lake
Churchill River
MAWDSLEY LAKE
Churchill River
Buffalo Narrows
Churchill River
0 20 40 60 80 100 Km

FIGURE 2 WATERSHED BOUNDARIES IN THE STUDY AREA

rapid fall between August and December. Daily temperatures fluctuate considerably. In the northerly areas, frost occurs approximately 60% of the year, and in all months except June and July. Ice thaw starts in May, and freeze-up usually occurs in November (Masterton et al. 1976). Most of northern Saskatchewan lies within the zones of discontinuous and continuous permafrost. In December, less than three hours of sunshine per day is recorded and the altitude of the sun is low; it is highest in June and July when hours of sunshine per day approach 20.

Average annual precipitation for northern Saskatchewan is approximately 460 mm, ranging from 362 mm (Uranium City) to 561 mm (Collins Bay). Of this, rainfall constitutes an annual average of 203 mm (Uranium City) to 382 mm (La Ronge); snowfall constitutes an annual average of 136.6 cm (137 mm rainfall equivalent) at Stony Rapids to 244 mm rainfall equivalent at Collins Bay. The average number of days with measurable rain and snow is 69 and 68, respectively.

Most precipitation falls during the summer months, due to the moist Pacific air mass. Snowfall accounts for the remaining two-fifths of the total annual precipitation. Snow cover is continuous for approximately six months, from late October until the middle of April or early May (Potter 1965).

Evaporation values estimated for northern Saskatchewan from aerial evaporation maps range from 50-60 cm.

The disperal ability of the atmosphere is a function of the wind speed, duration, direction, and atmospheric stability. The maximum average wind speeds observed at Cree Lake, Island Falls, La Ronge, and Uranium City usually occur during the warmer months from April to October as a result of increased frontal activity. Conditions of calm are more common during winter. The annual average wind speed is less than 16 km/h.

Conditions of calm and light winds with a duration of twenty-four hours or less occur with a frequency of approximately 5% in northern Saskatchewan. The maximum hourly wind speeds at La Ronge and Uranium City are 58 and 68 km/h, respectively. Prevailing winds are those from the west quadrant (SW-W-NW), although north quadrant winds (NW-N-NE) are almost as common.

Atmospheric stability describes the degree to which vertical motion is restricted and existing turbulence is suppressed. It is related to both wind speed and vertical temperature profile which in turn is a function of solar radiation. Instability occurs with high positive radiation (directed toward the ground) and low wind speed. Stability occurs with high negative radiation (night time conditions) and light winds.

Neutral conditions occur with cloudy skies or high wind speeds. Atmospheric stabilities are classified into seven categories, ranging from extremely unstable to extremely stable. Stable conditions are considered most limiting for atmospheric dispersal. This category occurs approximately 25% of the year in northern Saskatchewan.

3.2 Hydrology and Water Chemistry

The quantity and character of streamflow, or flux of water into and out of an area, are determined by rainfall, by evaporation and infiltration, by snowmelt, and by contribution from groundwater sources. There is a general lack of streamflow data for northern Saskatchewan.

The monthly median-yield streamflow for northern Saskatchewan was calculated from the available data to be 6.5 $L/s \cdot km^2$. This figure was based on data from the Inland Water Directorate.

The uranium properties lie either within the Lake Athabasca drainage basin which feeds into the Arctic Ocean via the Slave and Mackenzie Rivers, or within the Churchill River drainage basin which drains to Hudson Bay. In general, the waters in the Athabasca sandstone region are very soft. The concentration of total dissolved solids is below 20 mg/L. This is very low considering that a range of up to 1000 mg/L can normally be encountered in fresh waters. The concentrations of dissolved organic carbon fall within the normal range of 1-10 mg/L, often half the total dissolved component. In the Lake Athabasca region, values for total dissolved solids and conductivity are generally double those found on the sandstone.

Little is known about the origin, composition, and chemical nature of natural organic materials in surface waters. They consist principally of fulvic and humic acids, which are large molecular weight compounds, resistant to degradation by microorganisms. They are also complexing agents, forming stable metal humates and fulvates which are soluble in fresh water. Because of their predominance in the dissolved solids component of northern Saskatchewan waters, they can be expected to play a larger role in forming complexes with transition metals than in other areas where inorganic systems dominate. The lakes in northern Saskatchewan have near neutral to slightly acidic pH waters with very little buffering capacity. Some sediments form a thick organic ooze with a reducing environment. Both pH and Eh are important factors in controlling solubilization and agglomeration (and therefore sedimentation) in aqueous systems. Attractive forces between metals and clay minerals or hydrous oxides are strong in alkaline conditions. However, these forces decrease with pH because of the competition with hydrogen ions, resulting in the solubilization of metals. Lower redox potential favours the less soluble forms of uranium and vanadium, U(IV), and V(III); and the more soluble forms of iron and manganese, Fe(II) and Mn(II).

3.3 Regional Drainage Patterns

3.3.1 Cluff Lake Region - AMOK. The Amok Limited uranium-bearing property lies to the northeast of Cluff Lake. Bodies of water affected by the operation are Cluff Lake, Snake Lake, Island Lake and Boulder Creek in the Douglas River system. A flow schematic for the Cluff Lake area may be found in Figure 3.

Water quality parameters for several lakes in the Cluff Lake region are outlined in Table 12. These waters indicate levels of major cations and anions typical of most surface waters.

3.3.2 Key Lake - KLMC. The Key Lake Mining Company Limited deposits at Key Lake lie within the Wheeler River sub-basin of the Wollaston Lake watershed. The description of the regional hydrology is based upon information collected by Environment Canada on the Geikie River, and from stations recently constructed in the area.

There is a poorly developed surface drainage system in the area, typically consisting of lakes interconnected only at points of concentrated groundwater discharge. Seasonal flow peaks are generally low because of the modifying influences of interflow and lake storage. The average annual yield over the basin surface area is 0.2 m and represents approximately 45% of the average annual precipitation. Figure 4 is a flow diagram for the area.

The mean annual flow for the area over the 1966-77 period was 1.524×10^6 m^3, or 205 000 m^3/km^2. The average annual yield for the Key Lake area was therefore assumed to be 200 000 m^3/km^2. Frequencies of extreme yields were estimated, based upon the Geikie River record, to be 167 m^3 once in ten years, and 293 m^3 once in one hundred years (BEAK 1979).

Water Quality. The average total dissolved solids (TDS) value was 26 mg/L, the average total suspended solids (TSS) value was 6 mg/L, and conductivities ranged from 9-22 µS/cm. These levels were similar to those of Wollaston Lake and lakes in the Lac La Ronge area. Lakes in the Key Lake area had lower total dissolved solids than Cluff Lake and its associated streams which ranged from 65 to 100 mg/L (Amok 1976).

The bicarbonate anion was predominant over sulphate and chloride, a characteristic of shield lakes (Armstrong and Schindler 1971). The major cation was

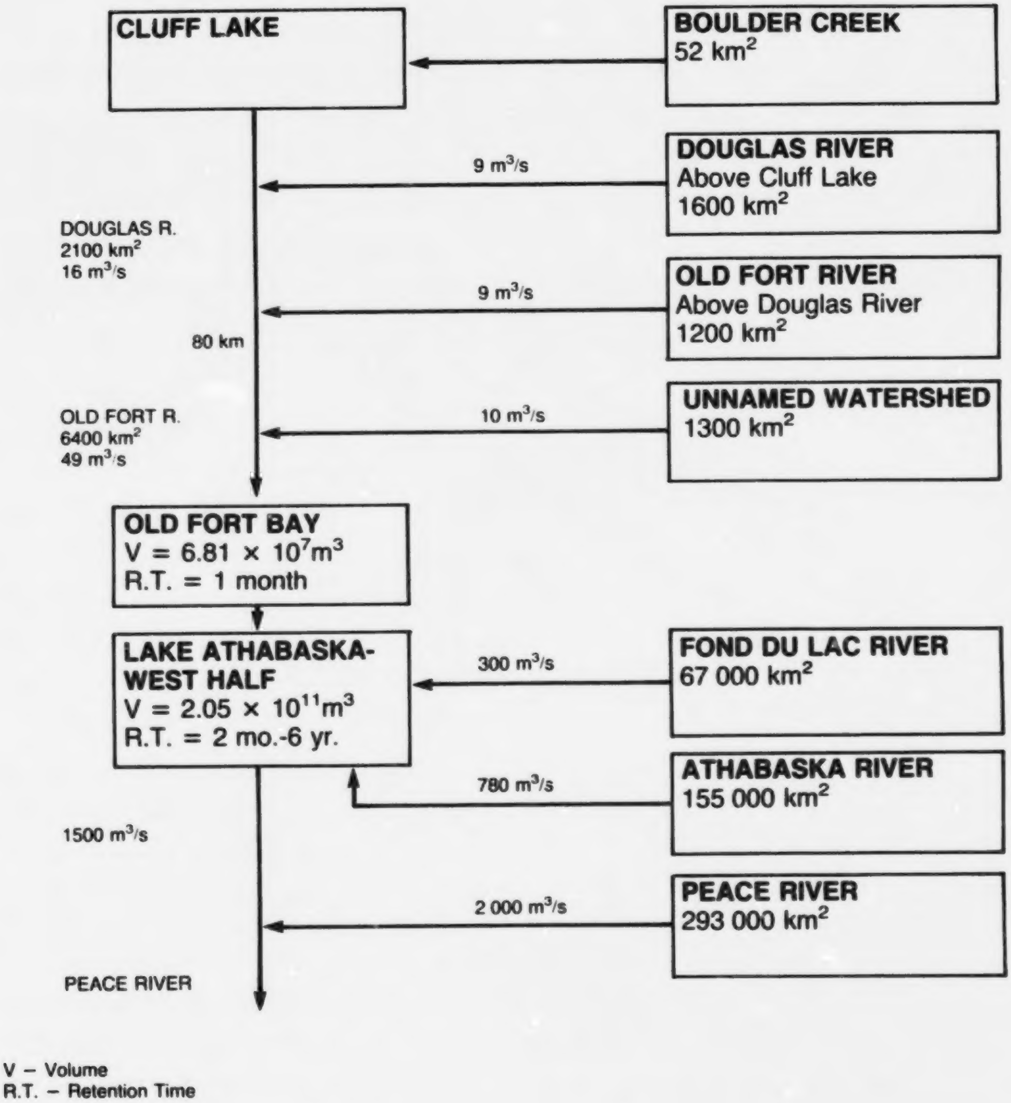
CLUFF LAKE
BOULDER CREEK
52 km²
9 m³/s
DOUGLAS RIVER
Above Cluff Lake
1600 km²
DOUGLAS R.
2100 km²
16 m³/s
9 m³/s
OLD FORT RIVER
Above Douglas River
1200 km²
80 km
OLD FORT R.
6400 km²
49 m³/s
10 m³/s
UNNAMED WATERSHED
1300 km²
OLD FORT BAY
V = 6.81 × 10⁷m³
R.T. = 1 month
LAKE ATHABASKA-
WEST HALF
V = 2.05 × 10¹¹m³
R.T. = 2 mo.-6 yr.
300 m³/s
FOND DU LAC RIVER
67 000 km²
780 m³/s
ATHABASKA RIVER
155 000 km²
1500 m³/s
2 000 m³/s
PEACE RIVER
293 000 km²
PEACE RIVER
V – Volume
R.T. – Retention Time

FIGURE 3 CLUFF LAKE DRAINAGE SYSTEM

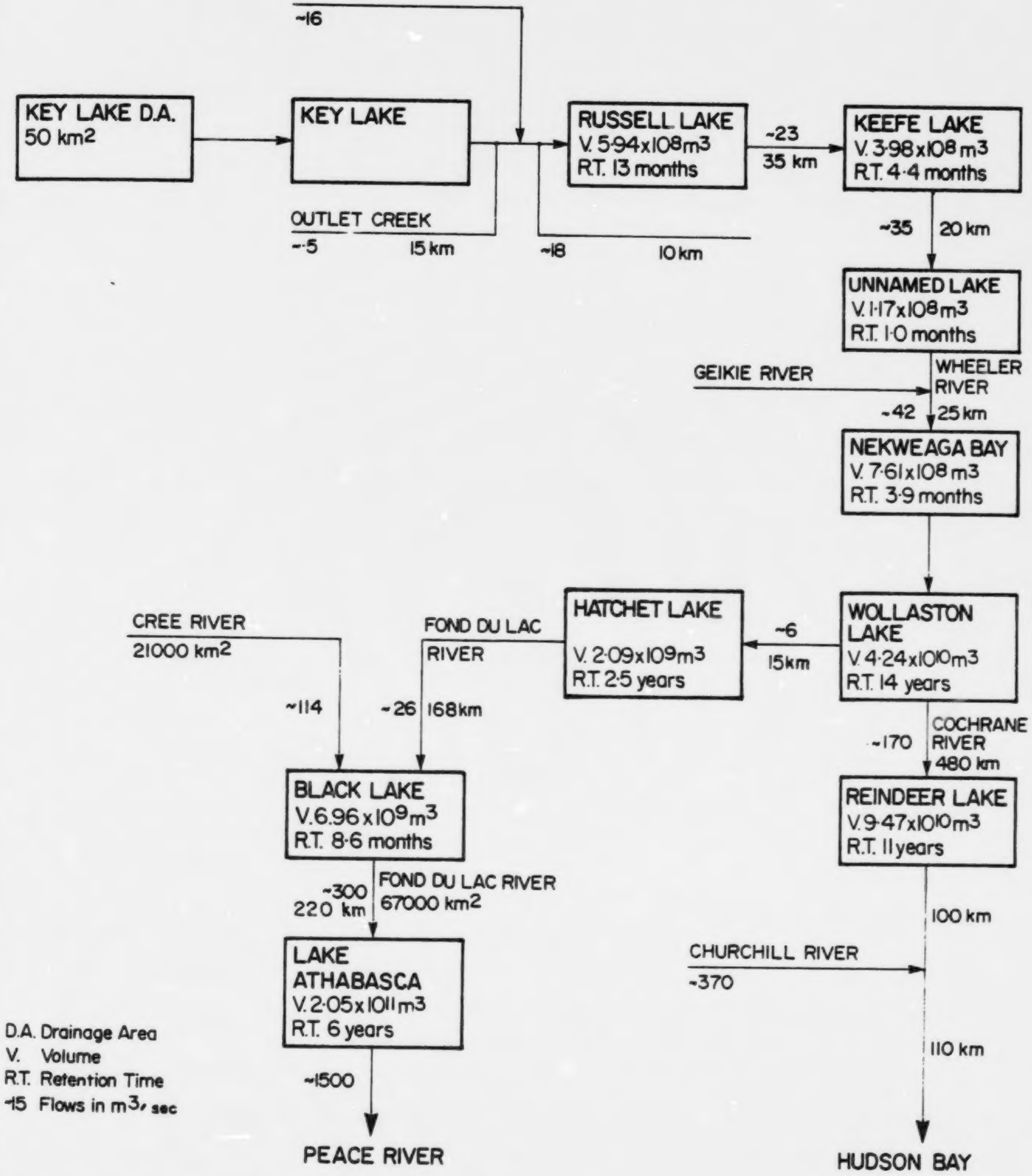
~16
KEY LAKE D.A.
50 km2
KEY LAKE
RUSSELL LAKE
V. 5·94x108m3
R.T. 13 months
~23
35 km
KEEFE LAKE
V. 3·98x108m3
R.T. 4·4 months
OUTLET CREEK
~5
15 km
~18
10 km
~35
20 km
UNNAMED LAKE
V. 1·17x108m3
R.T. 1·0 months
GEIKIE RIVER
WHEELER
RIVER
~42
25 km
NEKWEAGA BAY
V. 7·61x108m3
R.T. 3·9 months
CREE RIVER
21000 km2
FOND DU LAC
RIVER
HATCHET LAKE
V. 2·09x109m3
R.T. 2·5 years
~6
15km
WOLLASTON
LAKE
V. 4·24x1010m3
R.T. 14 years
~114
~26
168km
COCHRANE
RIVER
~170
480 km
BLACK LAKE
V. 6.96x109m3
R.T. 8·6 months
REINDEER LAKE
V. 9·47x1010m3
R.T. 11 years
~300
220 km
FOND DU LAC RIVER
67000 km2
100 km
CHURCHILL RIVER
~370
LAKE
ATHABASCA
V. 2·05x1011m3
R.T. 6 years
D.A. Drainage Area
V. Volume
R.T. Retention Time
~15 Flows in m3/sec
~1500
110 km
PEACE RIVER
HUDSON BAY

FIGURE 4 KEY LAKE DRAINAGE SYSTEM

TABLE 12 WATER QUALITY SUMMARIES OF THE CLUFF LAKE AREA

Parameter	MacFarlane River Station No. 00SA07MB001	Cluff Lake	Carswell Lake	Mill Lake	Island Lake	Douglas River Above Confluence
pH	8.0	8.08	8.10	7.20	7.02	7.40
Conductivty (µS/cm)	25	133	227	56	68	110
TDS	15	0	136	59	52	77
Total Hardness	12.1	66	114	26	37	43
Alkalinity	6	60	95	22	25	16
HCO_3		73	116	27	31	20
Cl		3	16	0.7	5	22
SO_4		1	1	1	1	1
Ca		14	26	8	6	10
Mg		8	12	2	5	4
Na		2.0	2.0	1.3	1.5	3.0
K		0.6	1.0	0.3	0.6	0.8
Organic C		8.8	10.2	20.0	16.4	8.8

All units except pH are mg/L except where stated otherwise.
References: Amok (1976); Inland Waters Directorate, Environment Canada (1977).

sodium at 1.0 mg/L, with potassium, calcium, and magnesium usually lower. Levels of total phosphate, orthophosphate, and total nitrogen were low. The mean concentration of organic carbon was 4.6 mg/L with a range 1 to 18 mg/L. Metals and trace elements were usually below the detection limit. Only iron was found in appreciable quantities in most water bodies.

Radionuclides were occasionally detected. Total Ra-226 ranged from below the detection limit in summer to a maximum of 0.12 Bq/L in March or October. Levels of Pb-210 and Po-210 were highest in Karl Ernst Lake with 0.36 and 0.37 Bq/L, respectively. Both were detected at other points in the area at concentrations in excess of 0.037 Bq/L. Uranium and thorium were usually undetectable, although Seahorse Lake waters did contain both. Uranium was also observed in Karl Ernst and Lower Seahorse Lakes, ranging from 1.1-5.1 µg/L (KLMC 1979).

3.3.3 Rabitt Lake Region - Eldor Resources. The uranium mine at Rabbit Lake, operated by Eldorado Resources, lies within the Wollaston Lake sub-basin. This is the headwaters basin for both the Cochrane River system to Hudson Bay and Fond du Lac River system to Lake Athabasca. The site's location and drainage into Hidden Bay suggest that the water in this system is probably associated with the Fond du Lac system.

Baseline water quality is presumed to be similar to that in Wollaston Lake at Ross Channel, as below:

pH	8.0
conductivity	20 µS/cm
TDS	15 mg/L
total hardness	10 mg/L
major anions and cations	1 mg/L

(Inland Waters Directorate, 1977d)

A flow schematic for the Rabbit Lake region, including drainage areas, flow patterns, and retention times may be found in Figure 5.

3.4 Aquatic Radiochemistry

Water radiochemistry has been studied for the Key Lake area (see Section 3.3 above) and for the Beaver Lake area (Dyck 1978).

Sediment Chemistry. Sediment metal contents are highly variable, but several conclusions have been suggested by the results of geochemical surveys in northern Saskatchewan (e.g., Hornbrook and Garrett 1976):

1) Background levels of uranium in lake sediments are 1-2 ppm on the Athabasca Formation and 3-4 ppm on the crystalline basement.
2) Lake sediments are enriched in uranium near uranium ore deposits.
3) The association of uranium with vanadium is dominant in sediments from lakes on the Athabasca Formation.
4) An association of uranium with nickel and copper is dominant in sediments from lakes on the crystalline basement.
5) An association of uranium with molybdenum is present only in the north, and with cobalt only in the south of the study region.

Dyck (1978) has summarized information about the radium content of sediments in northern Saskatchewan. In the Key Lake area, the arithmetic mean radium content of lake sediments is 0.14 Bq/g.

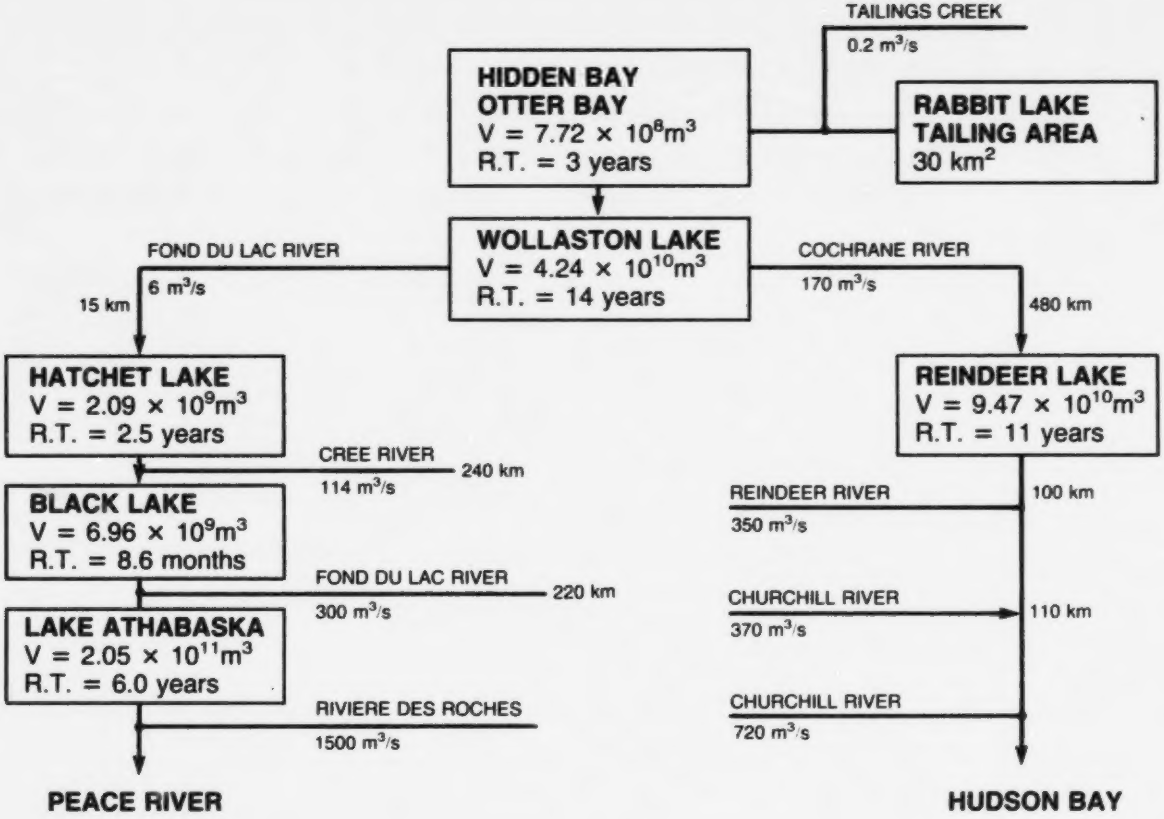
TAILINGS CREEK
0.2 m³/s
HIDDEN BAY
OTTER BAY
V = 7.72 × 10⁸m³
R.T. = 3 years
RABBIT LAKE
TAILING AREA
30 km²
WOLLASTON LAKE
V = 4.24 × 10¹⁰m³
R.T. = 14 years
FOND DU LAC RIVER
6 m³/s
15 km
COCHRANE RIVER
170 m³/s
480 km
HATCHET LAKE
V = 2.09 × 10⁹m³
R.T. = 2.5 years
REINDEER LAKE
V = 9.47 × 10¹⁰m³
R.T. = 11 years
CREE RIVER
114 m³/s
240 km
BLACK LAKE
V = 6.96 × 10⁹m³
R.T. = 8.6 months
REINDEER RIVER
350 m³/s
100 km
FOND DU LAC RIVER
300 m³/s
220 km
CHURCHILL RIVER
370 m³/s
110 km
LAKE ATHABASKA
V = 2.05 × 10¹¹m³
R.T. = 6.0 years
RIVIERE DES ROCHES
1500 m³/s
CHURCHILL RIVER
720 m³/s
PEACE RIVER
HUDSON BAY

FIGURE 5 RABBIT LAKE DRAINAGE SYSTEM

Adsorption Isotherms. There are processes operational in lake systems which are responsible for stripping both stable and radioactive elements from the water column. These processes, to a large degree, involve adsorption onto settling particles and adsorption directly onto sediments. A distribution between the solid and liquid phase may be described as the K_d value or distribution coefficient. K_d values used in the chain lake model are discussed in chapter six. The literature on adsorption isotherms or the way in which the distribution coefficient changes with concentration is sparse for uranium decay series radionuclides. The values which do exist for elements such as Ra-226 are mainly for mineral sediments and give values in the low hundreds. The higher number of reactive sites on the organic acids found in the lake sediments of northern Saskatchewan would suggest that the distribution coefficient for these sediments may be much higher. The experimental data collected in northern Saskatchewan by Key lake Mining Corporation (KLMC 1979) for lake water concentrations and sediment concentrations show a distribution between these two lake compartments in the mid-thousand range. Because of the non-equilibrium condition between lake water and interstitial sediment water, these values are expected to be upper limit K_d values for the organic sediments. The K_d values used in this study are listed in Appendix B.

3.5 Aquatic Ecology

Some fish are at the top of the aquatic food chain. Through the process of bioaccumulation they tend to concentrate some contaminants relative to water. The analysis of fish populations can serve as a long-term indicator of environmental quality.

More than thirty fresh water fish species are endemic to northern Saskatchewan (Scott and Crossman 1973). Four of the fish species which are of commercial and/or recreational importance are described below.

Lake Trout. These fish are important in commercial fisheries in the region. Lake Trout populations inhabit deep, cold water (Johnson 1971) and are usually found in bigger, deeper lakes of northern Saskatchewan. Lake trout are omniverous and feed on a wide variety of food items including crustaceans, aquatic insects, many species of fish, and even small mammals.

Throughout Canada, lake trout spawn mostly in October, but sometimes in September or November (Scott and Crossman 1973; Rawson 1961). Spawning primarily occurs on gravel or rubble substrates in shallow depths. The fertilized eggs, having diameters of five to six mm, fall into the crevices. Lake trout disperse into the lake after spawning.

Arctic Grayling. This species is an important sport fish. It is found only in cold freshwater drainages (Scott and Crossman 1973). Adults consume aquatic and terrestrial insects and a wide variety of invertebrates.

Spawning occurs in the spring at ice break-up, in small gravel or rock-bottomed streams. The egg diameter in the ovary is about 2.5 mm.

Lake Whitefish. Lake whitefish are widely distributed in cool northern fresh waters (Scott and Crossman 1973) and are a significant part of the commercial fishery in northern Saskatchewan. Adult lake whitefish are bottom feeders, their diet consisting mainly of benthic invertebrates and small fish (Rawson 1959; Johnson 1971; Scott and Crossman 1973).

Spawning occurs in shallow waters with gravelly or sandy substrates. Eggs, upon extrusion, measure about 2.3 mm in diameter but expand 30-40% after 24 hours in the water. Spawning is temperature dependent and, in northern Saskatchewan, occurs in late September to early October.

Walleye. These fish are important commercial and sport fish. Walleye prefer large, shallow, turbid lakes (Scott and Crossman 1973). Walleye feed throughout the water column on a diet from invertebrates to small fish.

Spawning occurs in spring or early summer on rocky areas below impassable falls, or on boulder to coarse-gravel shoals. Egg diameter is 1.5 to 2.0 mm.

Uptake of Radionuclides by Aquatic Biota. Chemical properties of radionuclides and background water quality affect the availability of radioisotopes for uptake. Physiological state and level of maturity of the individual fish also influence the uptake of radioisotopes. Therefore, without more specific water chemistry and fish physiological data for northern Saskatchewan lakes, little more than generalities can be discussed.

Uptake of trace elements by fish occurs either by ingestion or by active or passive absorption through the gills. The concensus from the literature is that the ingestion of contaminated food items is the major mechanism of uptake (Dvorak and Lewis 1978). The uptake of radionuclides varies according to the habitat and food habits of each species.

Radionuclides ingested or absorbed by fish are eventually returned to the sediments through defaecation (Parsont 1967) or decomposition after death. Fish may also transfer radionuclides to the terrestrial system by being consumed by piscivorous mammals, particularly man. The major radionuclides of concern in fish used for food are Ra-226, Pb-210, and Po-210. Thorium would be of concern in acidic waters (pH 5.5); however at higher pH, thorium is not of great concern because it is present predominantly as an insoluble hydroxide.

In aquatic systems, trace elements are partitioned between the water and the sediments which act as both a sink and a reservoir. Rooted vegetation and benthic invertebrates potentially accumulate trace elements from the water and sediments. Phytoplankton both adsorb and desorb trace elements from water. Grazers and low -order consumers appear to concentrate trace elements to the greatest degree, particularly those which are sedimentary or detrital feeders. These organisms are prey to higher trophic levels, and are the basis for radionuclide transfer through the food chain.

Because commercially important fish species spawn in either gravelly or cobble areas, or in weedy areas that are well flushed, the radiation dose to fish eggs is expected to be lower than in areas of fine silts and organic material that are often reported in the geochemical literature. Therefore, the average sediment level of radionuclides must be carefully interpreted when making dose calculations.

3.6 Terrestrial Ecology

The terrestrial food web is complex. For any given plant or animal species there is a wide range of consumers; for any consumer there is considerable range in diet. Almost all herbivores devour some animal matter, and carnivores rarely exclude plants from their diets. The food "chain" is thus better conceived of as a "food web". Typical food chains for northern Saskatchewan are listed in Table 13.

In calculating uptake to critical terrestrial biota (Sections 4 and 7), three species were chosen: moose, caribou, and beaver. Moose were presumed to eat grasses and herbs and aquatic vegetation during the snow-free months and shrubs during winter. The diet of caribou was conservatively estimated to be principally lichens with some grasses. Beaver were assumed to feed on aquatic plants and woody vegetation (Banfield, 1974).

3.6.1 Radionuclide Uptake and Cycling. Radionuclides taken up by plants may be ingested by herbivores and carnivores. Radionuclides may also be assimilated while breathing, grooming, or drinking water.

Uptake by plants varies with the physical and chemical state of the radionuclide, the mode of the impingement, and the plant part and species. Plant species show great differences in the means whereby, and the degree to which, they take up radionuclides. These differences in plant species have not been well investigated for radionuclides of the uranium decay series. Uptake factors are influenced by the following:

1) concentration factors are greater in nutrient-poor environments than in nutrient-rich environments (Ophel 1978; Polikarpov 1966; Vanderploeg et al 1975);
2) concentration factors are greater in freshwater than in salt water;
3) uptake is greater with animal species of restricted range, longer life spans, and higher metabolic rates;

TABLE 13 COMMON FOOD CHAINS IN NORTHERN SASKATCHEWAN

A. PLANT: HERBIVORE, OMNIVORE	
1. Grasses, Forbes	- snowshoe hare, vole, mouse, woodchuck, mule deer, caribou, moose
2. Seeds, berries, nuts	- chipmunk, squirrel, vole, mouse, fox, skunk, bear
3. Buds, bark	- snowshoe hare, woodchuck, squirrel, beaver, vole, mouse, porcupine
4. Aquatic vegetation	- beaver, muskrat, porcupine, moose
B. INSECT: INSECTIVORE AND OMNIVORE	
1. Invertebrates	- shrew, chipmunk, squirrel, muskrat, mouse, vole, bear, marten, fox, wolf, mink, skunk, otter
2. Fish	- otter, mink, wolverine, ermine, wolf
3. Shrew, mouse, vole	- fox, marten, fisher, ermine, mink, otter, hawk, owl, fish
4. Squirrel, chipmunk	- fox, wolf, bear, marten, fisher, ermine, skunk, lynx, hawk, owl
5. Beaver	- wolf, bear, fisher, wolverine, otter, lynx
6. Muskrat	- wolf, fox, fisher, mink, otter, lynx, hawk, owl
7. Snowshoe Hare	- wolf, fox, marten, ermine, mink, lynx, hawk, owl, man
8. Caribou	- wolf, wolverine, lynx, man
9. Moose	- wolf, man
C. CARNIVORE: HIGHER CARNIVORE	
1. Ermine	- fox, marten, fisher, wolverine, hawk, owl
2. Marten	- fisher, lynx, hawk, owl
3. Wolverine, otter, lynx	- wolf
4. Mink	- wolf, fox, fisher, hawk, owl

4) the degree to which a predator is exposed depends on which parts of the host are consumed. Most predators eat muscle and organ tissue. Hence, bone-seeking radionuclides such as Ra-226 would not be taken up to the same degree. Rather, soft tissue-seeking isotopes such as Po-210 would be taken up.

When the radiation dose to man is calculated, he is considered to be the predator of moose, caribou and beaver.

3.6.2 Rare and Endangered Species.

Flora. The distributional patterns of rare and endangered species of plants are not clearly established for northern Saskatchewan. It has been suggested that increased radiation levels might be a factor in harming rare species of small population by incurring genetic damage.

Rare species have been found near Cluff Lake. No rare plant species were recorded in the Key Lake vicinity.

Furbearers and Small Mammals. There are no officially endangered or regionally rare mammalian fur-bearing species in northern Saskatchewan. Several reports (BEAK 1979; Bayda et al. 1978) provide details on local status of furbearers.

Large Mammals. No species of large mammals is rare or endangered in northern Saskatchewan. Caribou is of greatest concern in northern uranium developments, principally because of the potential for increased hunting and harassment of the herds.

Lichens, a major component of caribou diet, are susceptible to significant uptake of radionuclides. Because of this, caribou were identified in this report as one of the critical species, and their uptake of radionuclides was calculated. If widespread areas of northern Saskatchewan are made accessible to human intrusion or if important lichen stands become unavailable to caribou, its populations will decline. One concentration of woodland caribou has been noted in the Key Lake area. Woodland caribou are of more concern in southern areas, while barren ground caribbou are of more concern in northeastern areas of northern Saskachewan.

Moose are evenly and sparsely distributed in northern Saskachewan. Because of its potential for being hunted and consumed by man, moose was considered as another critical species for which radionuclide uptake was calculated.

Black bear are sparse, hardy, and adaptable. They are not considered in this report.

Birds. The bald eagle is considered an endangered species in North America. Northern Saskachewan provides good habitat for the fish-eating raptor and densities are moderate to high. Because mining developments are small in size, they appear to displace few birds directly. However, the major effects are the impacts on the watersheds and the potential for uptake of toxic materials by the bald eagle. This eagle is at the end of a long aquatic food chain and could accumulate radionuclides to a considerable extent.

Other species of concern include white pelican and double-crested cormorant. These species commonly nest in the southern and western portions of northern Saskatchewan. As fish-eating species, they are also potentially subject to radionuclide uptake. At present, only the range of the white pelican overlaps with uranium development areas (Cluff Lake).

Waterfowl are of concern because of their aquatic habits and alignment with the aquatic food chains. They inhabit northern Saskatchewan during droughty conditions in southern Saskatchewan. There are few waterfowl staging areas recognized in northern Saskatchewan except for the Cumberland Delta area at the southeast corner of the Northern Administrative District (N.A.D.). Several other areas in the south of the N.A.D. are recognized as high concentration areas for white pelican and double-crested cormorant.

3.7 Regional Land and Water Use

Certain modes of land and water use can be expected to transport radionuclides. Typical land uses include limited agriculture, forestry, consumption of wild plants, consumption of drinking water, fishing, hunting and trapping.

Agriculture. There is virtually no agriculture in northern Saskachewan. Domestic garden vegetables may, however, be grown despite the short growing season. Vegetables most likely to be grown from seed in the north include special varieties of leaf lettuce, radish, beet, chard, and summer turnip. Other crops can be grown with aids such as cold frames and greenhouses. The potential uptake of radionuclides by such plants will, in most cases, be slight. Incorporation of radionuclides will be most marked from aerial sources. Large-leaved plants may be expected to have greater deposition potential. As few perennial food crops are likely, uptake from soil may not be as important. Uptake through roots is generally a longer-term consideration.

Forestry. Commercial forestry is not considered feasible in the areas around the mine sites studied. There is assumed to be no transport of radionuclides from forestry practices.

The susceptibilities of various northern tree species to external radioactivity is uncertain. Jack pine appears to be among the more sensitive (Bayda et al. 1978) and is among the most common species in northern Saskatchewan. Even so, radiation dose levels of at least 0.3 Gy/a are required to elicit growth alterations in this species (Ibid, 5.54). Such radiation rates were considered unlikely to occur in the areas surrounding the three mining operations.

Consumption of Wild Plants. A number of plants are or could potentially be consumed by people in northern Saskatchewan. These include various berries, fruit, and perhaps wild rice and other plants. Other plants occasionally utilized include labrador tea, limlock and wild mint (for teas). Bullrush, cattail, yellow waterlily, arrowhead, wild licorice, and wild onion roots are also eaten occasionally. The potential for radionuclide uptake by these plants is generally unknown, but it is assumed to be greater near ore bodies and mine sites. The above foods are probably not consumed in significant quantities on a continuing basis. They are not considered in dose calculations.

Use of Drinking Water. The potential for uptake of radionuclides in drinking water could be greater near ore zones and mine sites where water is obtained from sources affected by the mineralization or the process discharges.

Fishing. Fishing is undertaken both by residents and by tourists in northern Saskatchewan. The extent of this activity in northern Saskatchewan has been discussed by Balmer, Crapo and Assoc. Ltd. (1976). There are 38 northern fly-in camps and several fishing zones near potential existing developments. With an average of 12.9 days spent fishing per angler, the use of northern lakes totalled 272 316 angler-days (Balmer, Crapo and Assoc. Ltd. 1976).

Commercial Fishery. Lake whitefish, pickerel, lake trout and northern pike are the most important commercial fish species in northern Saskatchewan. In 1977 to 1978, lake whitefish represented 41.7% of the total poundage of fish taken. Pickerel and northern pike each represented approximately 20% of the total and lake trout approximately 14% in that year. In 1977-1978, 7.7 million pounds of fish were taken from northern Saskatchewan lakes. Most of the catch went to the U.S.A.

Sport Fishery. The major fish species sought by anglers are northern pike, pickerel, lake trout, arctic grayling, rainbow trout, and brook trout. The latter two fish species are not endemic to Saskatchewan and are stocked in the more southerly lakes in northern Saskatchewan. Based on the number of fish caught and retained by licensed anglers, it is estimated that non-residents caught approximately 570 000 fish of the major species in northern Saskatchewan, and residents about 480 000 fish in 1975.

Hunting. Hunting is carried out in northern Saskatchewan by residents and tourists who hunt moose, caribou, ducks, and geese. Other game occasionally taken include upland birds (predominantly grouse), snowshoe hare, deer, and bear.

Waterfowl. Hunting of waterfowl may be of importance locally at certain times but is not sufficiently productive to attract many non-resident hunters. Uptake of radionuclides by waterfowl is possible and, through consumption of such birds, radionuclides could pass to man. Very few data are available concerning local consumption rates of waterfowl among north Saskatchewan residents.

Game Mammals. The species of big game hunted in northern Saskatchewan are caribou, moose and black bear. Minor deer and wapiti hunting may be involved in southern portions as well. Except in areas primarily south of the Churchill River, sport hunting is not extensively pursued. Hunting by residents is commonly undertaken and some communities depend on such game as a major source of food in fall and winter.

Caribou is of greatest interest because of the efficiency of the lichen-caribou-man food chain in accumulating airborne radionuclides such as Cs-137, Sr-90, Pb-210 (Tuomen and Jakkola 1973). The factors which influence the food chain efficiency are:

1. lichen uptake of radionuclides enhanced by slow growth rate, long lifetime, high capacity to absorb nutrients from air, and the generally nutrient-deficient habitat of lichens;
2. degree to which lichens form a major component of the diet of caribou in winter;
3. consumption of caribou meat and edible organs by man.

Studies indicate that caribou do indeed take up significant quantities of radionuclides from lichen (Tuomen and Jakkola 1973). Polonium-210 in particular seems to be concentrated in caribou meat and transferred thereby to man.

In a study concerning resident hunting in Saskatchewan, the big game species preferred for hunting were listed in the following order: white-tailed deer, moose, elk, antelope, mule deer, bear, and caribou. Of these, caribou are expected to be the most critical because of the large lichen component of their diet.

Because there are many Treaty Indians in northern Saskatchewan, and because game hunted by them is not recorded, it is virtually impossible to establish sustainable yield data for big game or virtually any other game species in northern Saskatchewan.

Trapping. Several relatively minor pathways of radionuclide movement to man occur as a result of trapping. Some trappers eat the meat of some of the animals they trap. Those species would likely include snowshoe hare and beaver. The trapper may also hunt big game during trapping activities. Beaver harvest in areas near uranium deposits are very low and would not substantially contribute to a radionuclide pathway to man.

Snowshoe hare is also occasionally consumed by northerners, and its flesh could also provide a route of radionuclide transfer to man. This might only be significant in areas where browse eaten by hares contains elevated quantities of radionuclides. Such might occur near the mine fence, because hares have relatively small home ranges. Harvest and consumption levels of snowshoe hare are not known.

4 DOSE ASSESSMENT

4.1 Dosimetry

Radioactive materials released to waters and the atmosphere will lead to human exposure through ingestion of water and foods and from direct exposure to contaminated shoreline, soils, water, and air. Dosimetry calculations are based on ICRP 26 (1977) methodology. The effective dose equivalent for individuals, H_E, is defined as:

$$H_E = \sum_T W_T H_T + 0.01 H_{skin} \quad (4.1)$$

where H_T = dose equivalent in tissue T;

W_T = weighting factor for each tissue, representing the ratio of the stochastic risk from irradiation of the tissue, T, to that for the whole body when uniformly irradiated;

H_{skin} = the skin dose

The weighting factors W_T have been defined for all body organs and tissues.

The collective effective dose equivalent, S_E, is given by:

$$S_E = \int_0^\infty H_E N(H_E) dH_E \quad (4.2)$$

where $N(H_e)$ is the number of individuals receiving an effective dose equivalent in the range H_e to $H_E + dH_E$.

The collective effective dose equivalent commitment, S_E^C, is obtained by integrating the collective effective dose equivalent rate over all time, i.e.,

$$S_E^C = \int_0^\infty S_E (t) dt \quad (4.3)$$

If integration is limited in time, the resulting integral is called the incomplete collective effective dose equivalent commitment.

For internal exposures, S_E^C is more conveniently represented as:

$$S_E^C = \int_0^\infty \bar{H}_E \quad I.C (t)dt \quad (4.4)$$

where I = total intake rate of food, water, or air by the population (kg/a or m^3/a);

$C(t)$ = concentration of radionuclide in food, water or air (Bq/kg or Bq/m^3);

$\bar{H}_E$ = population-averaged effective dose equivalent per unit of radioactivity intake (Sv/Bq).

The quantity I.C (t) is the collective intake rate (Bq/a). $\bar{H}_E$ for the population is approximated by the 50-year integrated committed effective dose equivalent for adults, $H_{50,E}$, which is commonly tabulated for occupational exposures.

The incomplete collective effective dose equivalent commitment, which is the quantity evaluated in this study, is the sum of the effective dose equivalent commitments (integrated to a finite time T) received by all populations in all identified regions for all radionuclides and through all exposure pathways. To account for the risks from hereditary effects to populations beyond the first two generations, as well as fatal skin cancer, the total incomplete collective effective dose equivalent commitment, S_y, is calculated as:

$$S_y = S^C_E + 0.25\, S^C_{gonads} + 0.01\, S^C_{skin} \qquad (4.5)$$

4.2 Exposed Population Groups

Three population groups are considered for dose calculation purposes:

- the Critical Group: this is selected as the Wollaston Lake community.
- the Airshed Population: this population group consists of all persons who live within 80 km of any of the three mining operation sites considered.
- The Watershed Population: this includes all persons living along the waterways transporting aquatic radionuclide releases from the three mining/milling sites.

An annual dose is calculated fo the most highly exposed individual. This individual is considered to be a member of the 'critical group'. Collective annual population doses are calculated for the 'Airshed Population' and the 'Watershed Population'.

Critical Group. This group is the one most exposed to the radiological impacts of the mining operations. The Wollaston Lake community is selected as the critical group because:

- it is the population centre closest to any of the three mining operation sites, and
- liquid effluents from the Rabbit Lake and Key Lake mining and milling operations pass through into Wollaston Lake before discharge to the arctic and Hudson Bay drainage systems.

The most highly exposed individual is a hypothetical individual who:

- is an adult living permanently in the Wollaston Lake community,
- uses Wollaston Lake water for drinking, bathing, and all domestic needs,
- eats fish from the lake, game from the surrounding area, and vegetables from his own garden, and
- walks on the lakeshore and swims in the lake.

The above lifestyle habits are not expected of everyone living in the Wollaston Lake community; they are selected so as to avoid the possibility of underestimating the annual dose to the most highly exposed individual. For that reason, the quantities of local food consumed are chosen to be upper limits of consumption (See Section 5.1 of this report).

Airshed Population. This population is estimated at 679 people, and includes the population of the Wollaston Lake community. The average Wollaston Lake community resident will consume less contaminated food and water than the most highly exposed individual. For collective dose calculations, the Wollaston Lake members of the Airshed Population are assumed exposed to the same radionuclide pathways as the most highly exposed individual, but to a lesser degree. See Sections 5.2 and 6.2 of this report.

The collective dose to the remaining Airshed Population includes (a) external doses from air contamination and ground contamination, (b) internal doses from inhalation of radon and particles, and (c) internal doses from consumption of game and regional vegetables.

Watershed Population. This population group is estimated at 8059 people. They live in communities along the waterways which transport aquatic radionuclide releases. The collective dose to the Watershed Population includes (a) internal doses from ingestion of contaminated water and food, and (b) external doses from beach walking and immersion in water.

4.3 Radiological Impact from Atmospheric Emissions

4.3.1 External Dosimetry. External radiation dose to the local population and critical group was calculated through the summation of dose from a variety of sources including immersion in bath water and walking on ground containing radionuclides derived from mining and milling.

External gamma photon dose absorption rate factors in air were obtained from Kocher (1979). These factors were converted to effective dose equivalent by multiplying by 0.8 (CEC 1979). In the case of ground contamination, dose rate factors were calculated at a height of 1 metre above ground.

Electron dose absorption rate factors for skin were obtained from Kocher and Eckerman (1981). These factors were calculated at a depth of 70 μm below the body surface which corresponds to the depth at which radiosensitive tissues lie.

Total skin dose is the sum of the gamma dose (without the 0.8 correction factor referred to above) and the electron dose at 70 μm below body surface.

The whole body effective dose equivalent is the sum of the gamma dose (with the 0.8 correction factor applied) and 0.01 x skin dose. The factor 0.01 is the weighting factor for skin dose.

4.3.2 **Internal Dosimetry.** The internal radiation dose to the local population and critical group was calculated based upon summing the lung intake of radionuclides in gaseous and particulate forms and the intake of radionuclides ingested as food.

Effective dose equivalents per unit intake of radioactive material were obtained from ICRP-30 (1979) for adults. Inhalation rate for the adult was taken to be 8300 m^3/a (ICRP-23 1975). Internal dose conversion factors are given in Tables 14 and 15, and in Appendixes C and D.

The dose conversion factor for the inhalation of radon daughters was based on the regional lung dose concept (ICRP-32 1981) which splits up the weighting factor of 0.12 for total lung dose into 2 parts: 0.06 for the mean dose to the basal cell layer of the tracheobronchial region and 0.06 for the mean dose to the pulmonary region. The calculated effective dose equivalent for the inhalation of radon daughters has a range of values of $(0.42 \times 1.1) \times 10^{-2}$ Sv per WLM. A value of 0.01 Sv/WLM was selected for this study. The unit WLM refers to the exposure of occupational workers at a radon daughter concentration of 1 Working Level (WL) for 1 working month (M) of 170 working hours. One WL represents 1.3×10^5 MeV of alpha energy released from any combination of short-lived radon daughters in 1 litre of air during their decay to Pb-210. It is equivalent to 3.7 Bq/L of radon in equilibrium with its short-lived daughters.

Exposure of the public at 1 WL for one year (8760 hours), assuming the average inhalation rate for members of the public to be one-half of that of occupational workers during working hours (ICRP 23), is:

TABLE 14 CRITICAL GROUP DOSE CONVERSION FACTORS USED FOR CALCULATING DOSE TO MOST HIGHLY EXPOSED INDIVIDUAL

	Internal			External			
	Ingestion (Sv/Bq)		Inhalation (Sv/a per Bq/m^3)	Beach (Sv/a per Bq/kg)	Swim (Sv/a per Bq/kg)	Air Immersion (Sv/a per Bq/m^3)	Ground Contamination (Sv/a per Bq/m^3)
	Infant	Adult					
U-238	0.13E-06	0.620E-07	0.140	0.297E-07	0.124E-06	0.583E-07	0.491E-08
U-234	0.14E-06	0.650E-07	0.157	0.81E-09	0.621E-08	0.265E-08	0.629E-09
Th-230	0.54E-04	0.146E-06	0.583	0.864E-09	0.594E-08	0.260E-08	0.513E-09
Ra-226	0.12E-05	0.297E-06	1.77E-02	0.324E-05	0.702E-05	0.127E-07	0.634E-07*
Pb-210	0.70E-05	0.135E-05	2.9E-02	0.324E-08	0.265E-07	0.153E-07	0.248E-08
Po-210	0.80E-06	0.432E-06	1.80E-02	0.162E-10	0.351E-10	0.161E-10	0.311E-12
Rn-222			3.38E-03			0.332E-05	0
Rn Daughters (WL)			6.75				

* Includes Rn.

TABLE 15 POPULATION DOSE CONVERSION FACTORS USED FOR CALCULATION OF COLLECTIVE DOSES TO AIRSHED AND WATERSHED POPULATIONS

	Internal		External			
	Ingestion Adult Only (Sv/Bq)	Inhalation (Sv/a per Bq/m^3)	Beach (Sv/a per Bq/kg)	Swim (Sv/a per Bq/kg)	Air Immersion (Sv/a per Bq/m^3)	Ground Contamination (Sv/a per Bq/m^3)
U-238	0.622E-07	0.140	0.378E-07	0.124E-06	0.583E-07	0.491E-08
U-234	0.703E-07	0.157	0.102E-09	0.783E-07	0.265E-08	0.629E-09
Th-230	0.146E-06	0.583	0.108E-08	0.756E-08	0.260E-08	0.513E-09
Ra-226	0.378E-06	1.77E-02	0.405E-05	0.891E-05	0.127E-07	0.634E-07*
Pb-210	0.135E-05	2.9E-02	0.405E-08	0.324E-07	0.153E-07	0.248E-08
Po-210	0.432E-06	1.8E-02	0.203E-10	0.432E-10	0.161E-10	0.311E-12
Rn-222		3.38E-03			0.332E-05	0
Rn Daughters (WL)		6.75				

* Includes Rn.

$$\frac{8760}{170} \times \frac{1}{2} = 25 \text{ WLM}$$

Thus, the dose conversion factor for radon daughters is:

$$1 \text{ WL} = 25 \text{ WLM/a} = 0.25 \text{ Sv/a}$$

This is the factor used for calculating doses from exposure to radon daughters outdoors. Assuming an equilibrium factor of 0.5 between radon and its daughters, the dose conversion factor for exposure indoors is:

$$0.037 \text{ Bq/m}^3 \text{ Rn-222} = 5 \times 10^{-6} \text{ WL} = 1.25 \times 10^{-6} \text{ Sv/a}$$

If at a given location, the concentration of radon daughters is X WL and the concentration of Rn-222 is Y Bq/m^3, and if the average individual spends 60% of the time indoors, then,

$$\text{dose rate} = (0.2X)(0.25) + (0.6Y)(1.25 \times 10^{-6}) \text{ Sv/a} \qquad (5.6)$$

4.3.3 Calculation of Annual Collective Doses. Collective inhalation and external doses were calculated for identified population locations making up the Mining Area Population. The doses were calculated from air concentrations, ground contamination levels, and the sizes of populations at these locations, as follows:

$$S_E^C = \sum_{ijk} C_{ijk} D_{ik} P_j K_E \qquad (5.7)$$

where:

S_E^C = collective effective dose equivalent commitment for inhalation + external exposure (Sv).

C_{ijk} = concentration of radionuclides i at location j through exposure pathway k (Bq/m^3 or Bq/m^2 depending on exposure pathway);

D_{ik} = dose conversion factor for radionuclide i and exposure pathway k;

P_j = population size at location j (assumed not to change with time);

K_E = exposure constant, related to percentage of time exposed, and ratio of time indoors and outdoors;

4.3.4 Calculation of Doses to the Most Highly Exposed Individual. The most highly exposed individual was assumed to be an adult. The inhalation rate of infants ($3.8 \text{ m}^3\text{/d}$) is about 6 times lower than that of adults ($23 \text{ m}^3\text{/d}$), whereas the dose per unit intake for the infant is less than a factor of 5 higher than that for adults. Also, adults consume substantially more vegetables and meat than do infants.

4.4 Radiological Impact from Aquatic Releases

4.4.1 Dose Factors for Exposure to Contaminated Shoreline Sediment. It is assumed that exposed shoreline sediment is uniformly contaminated to some depth. The air dose above a uniformly distributed monoenergetic gamma source in soil, distributed down to a depth which is infinite with respect to gamma attenuation and build-up, has been calculated by Beck and dePlanque (1968). The calculated dose rates are relatively insensitive to soil composition and moisture content and change very little in the first 10 metres above ground. Table 17 presents external gamma dose conversion factors at 1 metre above ground for radionuclides of interest, based on Beck's calculations and using radionuclide gamma energy and abundance data compiled by D.C. Kocher (1977). The bulk sediment density assumed in the calculations is 1.6 g/mL.

4.4.2 Dose Factors for Immersion in Contaminated Water. Water immersion dose conversion factors for swimming doses have been calculated by D.C. Kocher (1979), and by D.C. Kocher and K.F. Eckerman (1981).

4.4.3 Internal Dosimetry. Internal dosimetry is based on the same considerations and dose conversion factor data described in Section 4.3.

4.5 Dose Conversion Factors - Human Exposure

Dose conversion factors used in this study for external and internal exposures for each parent plus daughters in equilibrium are summarized in Table 14 and 15. Table 14 gives dose conversion factors used in calculating critical group doses. Table 15 gives dose conversion factors used in calculating collective doses.

The dose factors for external exposure given for the radionuclides include the contribution from the relatively short-lived daughters with which they are in equilibrium. Dose factors for shoreline exposure consider that exposed shoreline sediment is uniformly contaminated to some depth. They include corrections for shore width; they are based upon the air gamma dose at one metre above sediment and are corrected to effective dose equivalent. The external dose conversion factor for contaminated water is labelled 'Swim' under the Critical Group and Population Group categories (Sv/a per Bq/L). The skin dose is based upon the sum of photon and electron doses at 70 μm below the body surface.

For internal exposure to radionuclides, the Committed Effective Dose Equivalent conversion factors are given for infants and adults of the Critical Group and for adults of the Population Group. These factors are calculated for intake integrated over a period of 50 years after uptake. Gonadal dose factors, available only for Ra-226, are included as per equation 5.5, for adults of the Population Group.

Use of these dose conversion factors, together with the consumption rate of foods (internal exposure) and the exposure rate to contaminated sediment and water (external exposure) given previously, permit calculation of the internal and external dose rates (e.g., Sv/a) for the most highly exposed member of the Critical Group and the Airshed and Watershed Population Groups. Integration of the effective dose equivalent commitment rate from all radionuclides through all exposure pathways over a fifty year period gives the annual, incomplete collective effective dose commitment for that population. This quantity is hereafter abbreviated and called the collective population dose or the population dose.

4.6 Doses to Biota

Dose estimate calculations for biota are mainly performed using methodologies recommended in IAEA No. 172 (1976) and IAEA No. 190 (1979). Modifications to these methodologies were made where the methodologies were not directly applicable, or where necessary data were lacking.

Calculated doses to biota should be regarded as estimates only. Lack of data and of detailed methodology prevent the making of accurate dose calculations. A significant difference in the methodology for calculating doses to humans and to other biota is the fact that relative biological effectiveness weightings are not used for alpha and beta radiations in calculating doses to natural populations. Conservative exposure assumptions have been used, so that exposure estimates will be upper limits. Available data have been used, combined with reasonable suppositions about diet, behaviour patterns, and metabolic parameters.

4.6.1 Terrestrial Biota - General Considerations. Terrestrial biota include both plants and animals. Detailed information on dosimetry calculations is given in Appendix C. Calculations are summarized in Section 7 of this report.

Plants. Except for lichens, only external gamma radiation doses are calculated for terrestrial plants. Dose contributions are calculated for airborne and ground contamination. Internal plant doses, and beta doses from plant surface contamination are not calculated; the many uncertainties give little value to such calculations.

Lichens are a special case. They are long-lived plants which take their nutrition mostly from the air. They are known to concentrate radionuclides from the air

(UNSCEAR 1977; Persson 1971, 1972 and 1973). Doses are calculated for external radiation, as well as for alpha, beta, and gamma radiations from internal uptakes from air.

Animals. Both internal and external doses are calculated for three animals: moose, caribou, and beaver.

External doses are calculated for gamma radiation only, from air and ground contamination. No skin doses are calculated; the uncertainties in hide absorption and fur/hide contamination levels give little meaning to such calculations.

Internal doses are calculated based on dietary uptakes; radionuclide concentrations are calculated in the terrestrial and aquatic plants which animals eat. Dose contributions are calculated for ingested alpha, beta, and gamma emitters. No respiration doses are calculated because of the lack of animal lung dosimetry data.

4.6.2 Aquatic Biota - General Considerations. Aquatic biota include fish, fish eggs, plankton, molluscs, crustaceans, and aquatic plants. Detailed information on dosimetry calculations is given in Appendix 5. For all aquatic biota, the following dose contributors are considered:

- internal doses from incorporated radionuclides - doses from alpha, beta and gamma emissions are calculated;
- external doses from radionuclides in surrounding water - beta and gamma radiation is considered;
- external doses from contaminated sediment.

Not all of the above exposure sources affect every species.

4.6.3 Internal Dose Calculations - All Biota. Dose conversion factors for biota are not available. Calculations are based on geometry, radionuclide uptakes, and type and energy of radiations emitted by the radionuclides. Internal dose rates to the bulk of the organism are all calculated with the following equations:

Alpha Radiation:

dose rate = $(N\alpha)\ 5.76\ E\ \ C \times 10^{-4}$ μGy/h

where: $N\alpha$ = a factor dependent on the size of the organism. $(N\alpha)$ is 1 for all organisms except plankton for which $(N\alpha)$ is 0.3

C = concentration of radionuclide in organisms in mBq/g

E_α = alpha energy per disintegration

Beta Radiation:

dose rate = $(N\beta)\ 5.76\ E\ \ C \times 10^{-4}\ \mu Gy/h$

where: $N\beta$ = a factor dependent on the size of the organisms. $(N\beta)$ is 1 for all organisms except plankton for which $(N\beta)$ is 7.0×10^{-3}

Gamma Radiation

dose rate = $\Gamma\ P\ \bar{g}\ C \times 2.7 \times 10^{-7}\ \mu Gy/h$

where: Γ = a specific gamma ray constant tabulated for the gamma emissions of each radionuclide

P = tissue density (generally 1 g/cm^3)

C = radionuclide concentration in the organisms in mBq/g

$\bar{g}$ = a geometric factor, depending on the size and geometry of the organisms

4.6.4 External Doses to Aquatic Biota. Both beta and gamma emitters in water and in sediment can deliver external doses to aquatic biota. Alpha radiations do not have sufficient penetrating power to deliver external doses, except to plankton species from emitters in the water.

Doses to fish eggs are a special case. They are calculated according to data and methodology by Woodhead (1970). Other doses are calculated using published data and methodology. Full details are given in Appendix D and a summary of calculations and methodology is given in Section 7 of this report.

5 RADIOLOGICAL IMPACT OF ATMOSPHERIC EMISSIONS

This section presents the results of calculations of radiological impacts of atmospheric emissions. The significance of each calculated impact is discussed. A brief review of calculation methodology is given in each case.

Calculations were made for the following atmospheric radiological impact categories:

- annual dose equivalent to most highly exposed individual in the Critical Group - the Wollaston Lake community (see Section 5.1 below);
- annual collective dose equivalent to the Airshed Population - all persons living within 80 km of any mining operation site (see Section 5.2 below);
- levels of radioactivity in air and deposited on the ground from the air (see Section 5.1 below).

Complete details of calculations are given in Appendix C.

The total doses from both atmospheric and aquatic sources are summarized in Table 5.1.

5.1 Environmental Radionuclide Concentrations - Atmospheric Pathways

Three sets of radionuclide concentrations were calculated:

- radon in air
- particulates in air
- ground deposition of radionuclides

Calculations assumed that all mining/milling sites had been in operation for 15 years.

For the most part, all three concentrations are hardly distinguishable from background levels in the area. Atmospheric background levels recorded for this area were measured in 1981-82 at Dawn Lake (MacLaren 1982). They were:

airborne radon 0.02 - 0.2 Bq/m^3
airborne particulates 0.18 x 10^{-2} Bq/m^3

5.2 Most Highly Exposed Individual - Annual Dose Equivalent

The annual dose equivalent from atmospheric sources was:

Internal:	0.341E-1 mSv/a
External	0.226E-2 mSv/a
Total dose	0.364E-1 mSv/a

Table 16 presents a complete breakdown of these doses for each radionuclide.

TABLE 16 ATMOSPHERIC PATHWAY - DOSES TO MOST HIGHLY EXPOSED INDIVIDUAL (μSv/a)

	Inhalation		Ingestion	External		
	Radon	Particulate	Caribou	Air Contamination	Ground Contamination	Total
U-238	-	0.430	0.159E-1	0.179E-6	0.400E-1	0.486
U-234	-	0.480	0.179E-1	0.810E-8	0.500E-2	0.503
Th-230	-	1.03	0.124E-1	0.449E-8	0.300E-2	1.05
Ra-226	-	0.300E-1	0.807	0.211E-7	0.370	1.21
Pb-210	-	0.500E-1	0.173	0.252E-7	0.100E-1	0.264
Po-210	-	0.300E-1	5.74	0.268E-10	0.200E-3	5.77
Rn-222	0.252E+2	2.05	6.77	1.80	0.459	0.363E+2
Total	0.252E+2	2.05	6.77	1.80	0.459	0.363E+2

The annual dose of 36 μSv/a represents less than 1% of annual allowable dose to members of the general public. The annual dose was calculated by assessing the dose attributable to each appropriate diet and lifestyle habit. For doses from various sources, these are:

respiration volume	8 300 m^3/a (ICRP 23 1975)
garden vegetable consumption	50 kg/a
caribou consumption	50 kg/a
beaver consumption	10 kg/a
moose consumption	1 kg/a
time spent indoors	60% of time

Food consumption shown is deliberately set at a high level to avoid underestimating the annual dose rate to the most highly exposed individual. Beaver consumption is included to reflect the fact that the most highly exposed individual and some members of the critical group may consume aquatic mammals such as beaver and muskrat from watersheds downstream of a mining and milling operation. Moose consumption is probably not significant but is included here for completeness since there are moose in the Wollaston Lake area.

Inhalation Doses

annual dose = (annual resp. volume) x (air conc. or radionuclide)
x (dose conversion factor)

- Inhalation doses are calculated for radon and daughters and inhaled particulates. Dose from each particulate radionuclide is calculated separately.
- Radon inhalation doses are a special case - see Section 4.3.1 of this report.

Doses From Food Ingestion. For each food, the annual dose is: (food consumption kg/a) x (radionuclide conc. in food Bq/kg) x (dose conversion factor Sv/Bq ingested).

- Garden vegetables absorb radionuclides deposited onto ground and contain radionuclides deposited on leafy parts.
- Beaver and caribou have predominantly land plant diets. Caribou eat mostly lichen.
- Beaver eat some aquatic plants. Both beaver and caribou are assumed to drink contaminated water. The total radionuclide uptake of the animals is calculated before calculating food ingestion doses to man. The human dose contributions from eating caribou (Table 16) are pro-rata portions of the animal's total uptake for uptakes from atmospheric sources.

External Exposure

a) annual air contamination dose = (air concentration Bq/m^3) x (dose conversion factor Sv/a per Bq/m^3) x (0.6 x building shielding factor + 0.4)

- The figure 0.6 represents the 60% of time spent indoors, with partial shielding by the building. The remainder of the time (0.4 - 40%) is spent outdoors with no building shielding. As a conservative assumption in the present study, the building shielding factor was set equal to one.
- The calculation was done for each airborne radionuclide.

b) ground contamination dose = (ground contamination Bq/m^2) x (dose conversion factor Sv/a per Bq/m^2)

5.3 Collective Dose Commitment - Atmospheric Impact

Collective annual doses are:

Airshed Population:	Internal	0.235E-1 person-Sv/a
	External	0.158E-2 person-Sv/a
Total Collective Dose		0.250E-1 person-Sv/a

The collective dose consists of:

- dose received through the year from external sources.
- dose commitments from radionuclide uptakes received during the year. Collective doses are listed in Table 17 for each radionuclide.

TABLE 17 ATMOSPHERIC PATHWAY - COLLECTIVE DOSES TO AIRSHED POPULATION (person Sv/a)

		Internal		External		
Nuclide	Radon	Inhalation Particulate	Ingestion Caribou	Air Contamination	Ground Contamination	Total
U-238	-	0.302E-3	0.160E-4	0.138E-9	0.328E-4	0.351E-3
U-234	-	0.346E-3	0.160E-4	0.617E-11	0.410E-5	0.366E-3
Th-230	-	0.720E-3	0.106E-4	0.360E-11	0.224E-5	0.732E-3
Ra-226	-	0.202E-4	0.106E-4	0.155E-10	0.298E-2	0.329E-3
Pb-210	-	0.331E-4	0.122E-3	0.183E-10	0.895E-5	0.164E-3
Po-210	-	0.202E-4	0.514E-2	0.200E-13	0.149E-6	0.516E-2
Rn-222	0.167E-1	-	-	0.121E-2	0.269E-4	0.179E-1
Total	0.167E-1	0.144E-2	0.532E-2	0.121E-2	0.373E-3	2.50E-1

For collective dose calculations, the Airshed Population includes:

- the population of the Wollaston Lake community
- all other residents and transients inside the 80 km radius circles around the mining/milling operations sites.

Residents and transients in the Airshed Population are assumed to have lifestyles similar to Critical Group members, but neither they nor their game are assumed to drink contaminated water.

Calculation. Arithmetic summation of internal collective doses is simply done by multiplying intake by appropriate dose conversion factor as in Section 5.1 above. External doses are calculated for air and ground exposure. MILDOS predictions are used to determine air and ground radionuclide concentrations at all population centres.

6 RADIOLOGICAL IMPACT OF AQUATIC RELEASES

This section presents the results of calculations of radiological impacts of aquatic releases. The significance of each calculated impact is discussed. A brief review of calculation methodology is given.

Calculations were made for the following aquatic radiological impact categories:

- annual dose to the most exposed individual of the Critical Group - Wollaston Lake community (see Section 6.1 below);
- annual collective dose for the Watershed Population (see Section 7.2 below);
- levels of radioactivity in affected waters and in the sediments of those waters (see Section 7.3 below).

Complete details are given in Appendix D. The total doses from both atmospheric and aquatic sources are given in Table S.1.

6.1 Environmental Radionuclide Concentrations - Aquatic Pathways

Two sets of radionuclide concentrations were calculated:

- radionuclides in water
- radionuclides in lake sediments

Calculations assumed that all mining/milling sites had been in operation for 15 years.

Figures 6 and 7 show water level concentrations for the drainage basins for each site. In each figure, water concentrations are the incremental concentration increases caused by operation of the site named in the figure. Figures 8 and 9 show sediment concentrations of radionuclides for the drainage basins of each site.

For the most part, the variation in water sediment concentrations will be indistinguishable from normal background levels in the area. Complete details of aquatic modelling calculations are given in Appendix D.

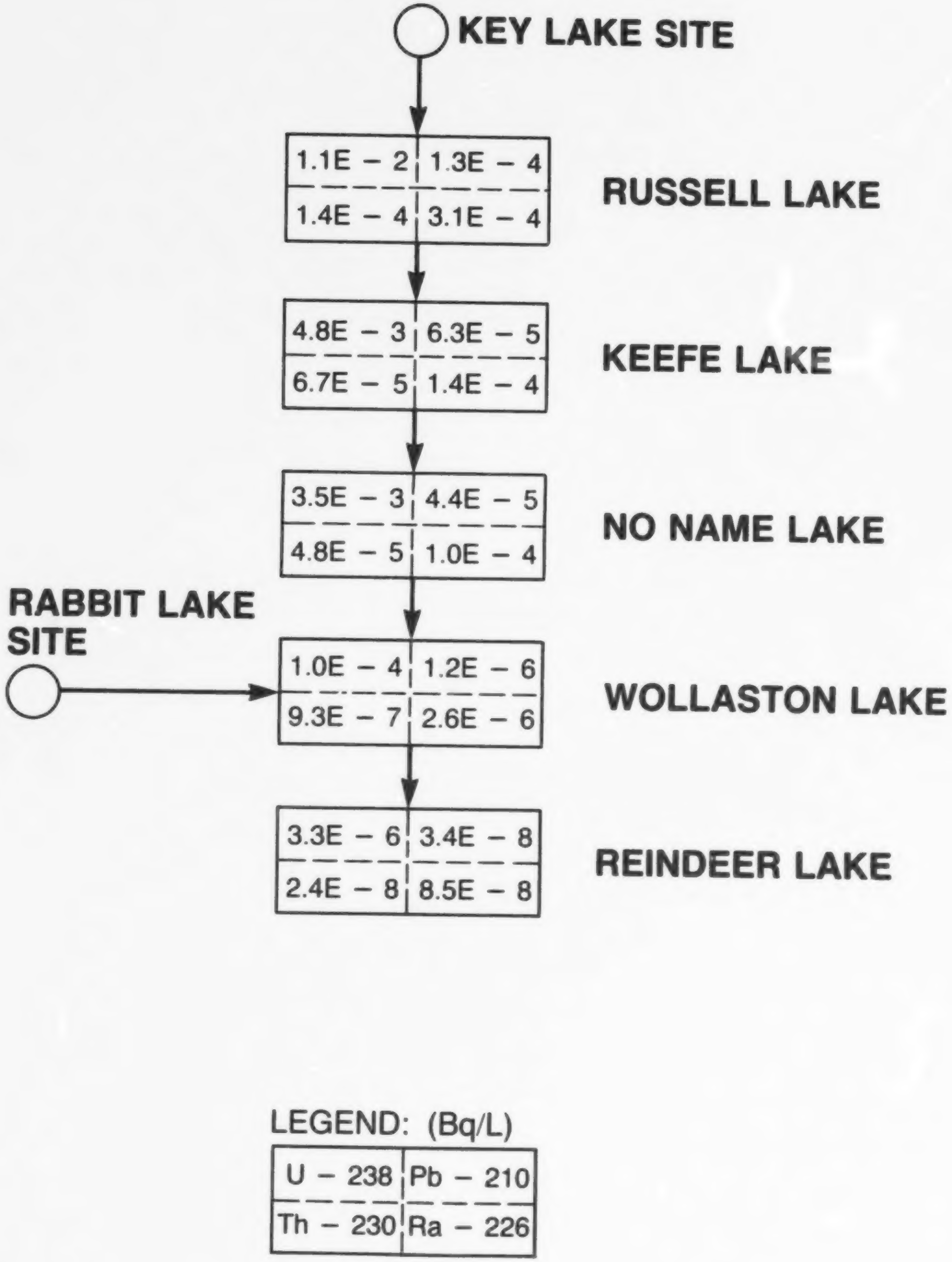
KEY LAKE SITE
1.1E – 2
1.3E – 4
1.4E – 4
3.1E – 4
RUSSELL LAKE
4.8E – 3
6.3E – 5
6.7E – 5
1.4E – 4
KEEFE LAKE
3.5E – 3
4.4E – 5
4.8E – 5
1.0E – 4
NO NAME LAKE
RABBIT LAKE SITE
1.0E – 4
1.2E – 6
9.3E – 7
2.6E – 6
WOLLASTON LAKE
3.3E – 6
3.4E – 8
2.4E – 8
8.5E – 8
REINDEER LAKE
LEGEND: (Bq/L)
U – 238
Pb – 210
Th – 230
Ra – 226

FIGURE 6 RADIONUCLIDE CONCENTRATIONS IN SURFACE WATERS DRAINING INTO REINDEER LAKE

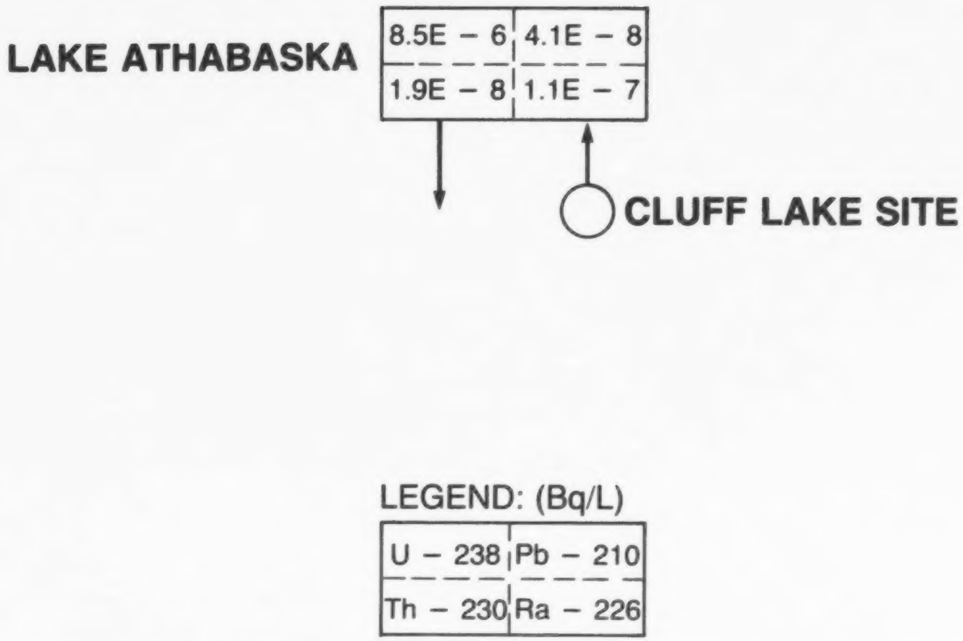
LAKE ATHABASKA
8.5E – 6
4.1E – 8
1.9E – 8
1.1E – 7
CLUFF LAKE SITE
LEGEND: (Bq/L)
U – 238
Pb – 210
Th – 230
Ra – 226

FIGURE 7 RADIONUCLIDE CONCENTRATIONS IN SURFACE WATERS DRAINING INTO LAKE ATHABASKA

6.2 Most Highly Exposed Individual - Annual Dose Equivalent

The annual dose from aquatic sources was:

Internal	0.290E-1 µSv/a
External	0.124E-4 µSv/a
Total Dose (Aquatic Sources)	0.290E-1 µSv/a

Table 18 presents a complete breakdown of these doses for each radionuclide. Internal doses are from water and fish consumption. External doses are from immersion (swimming, bathing) in contaminated lake water, and from walking on beaches containing contaminated sediment.

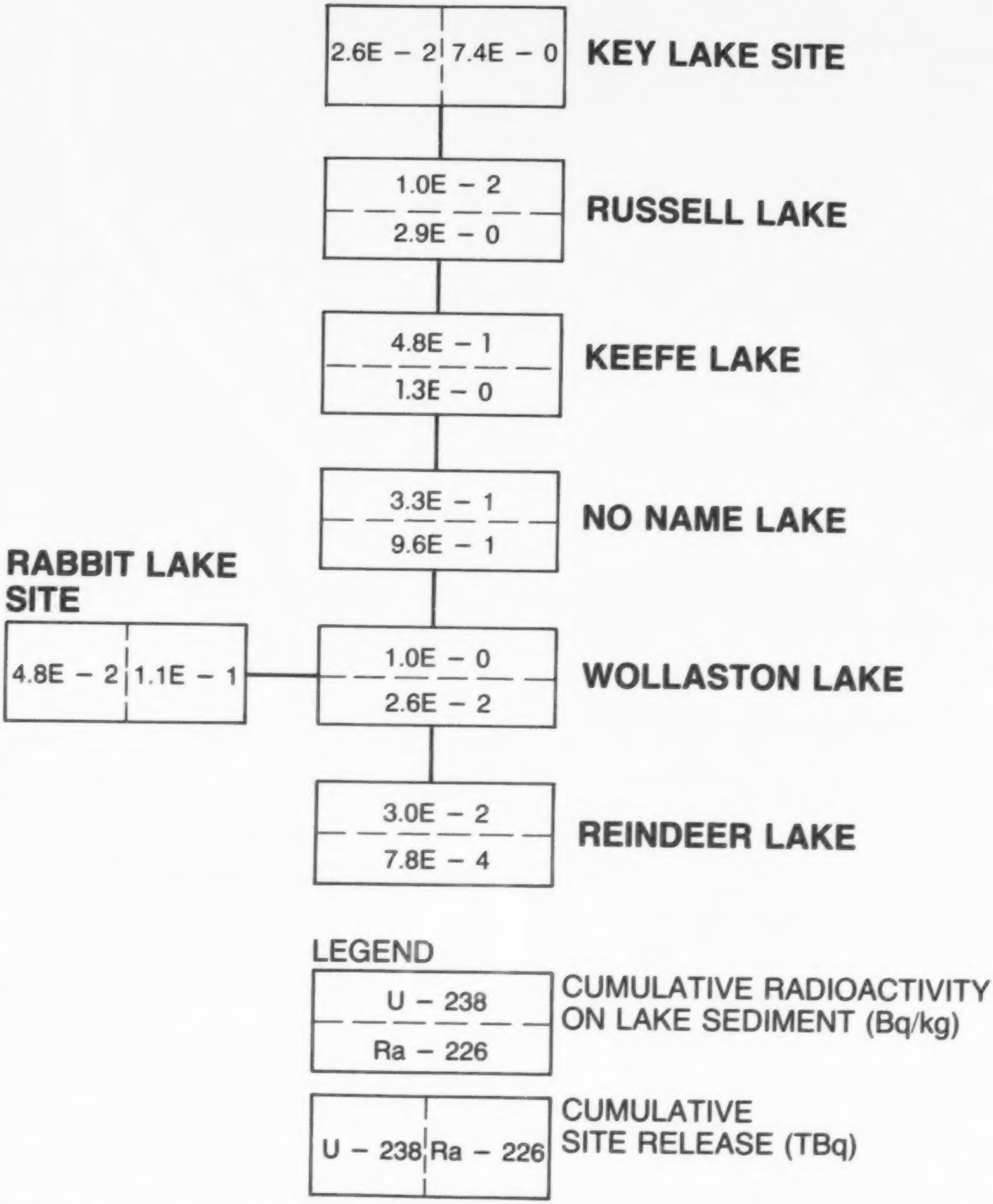
2.6E – 2
7.4E – 0
KEY LAKE SITE
1.0E – 2
2.9E – 0
RUSSELL LAKE
4.8E – 1
1.3E – 0
KEEFE LAKE
3.3E – 1
9.6E – 1
NO NAME LAKE
RABBIT LAKE SITE
4.8E – 2
1.1E – 1
1.0E – 0
2.6E – 2
WOLLASTON LAKE
3.0E – 2
7.8E – 4
REINDEER LAKE
LEGEND
U – 238
Ra – 226
CUMULATIVE RADIOACTIVITY ON LAKE SEDIMENT (Bq/kg)
U – 238
Ra – 226
CUMULATIVE SITE RELEASE (TBq)

FIGURE 8 RADIOACTIVITY DEPOSIT IN SEDIMENTS OF WATERSHEDS DRAINING INTO REINDEER LAKE

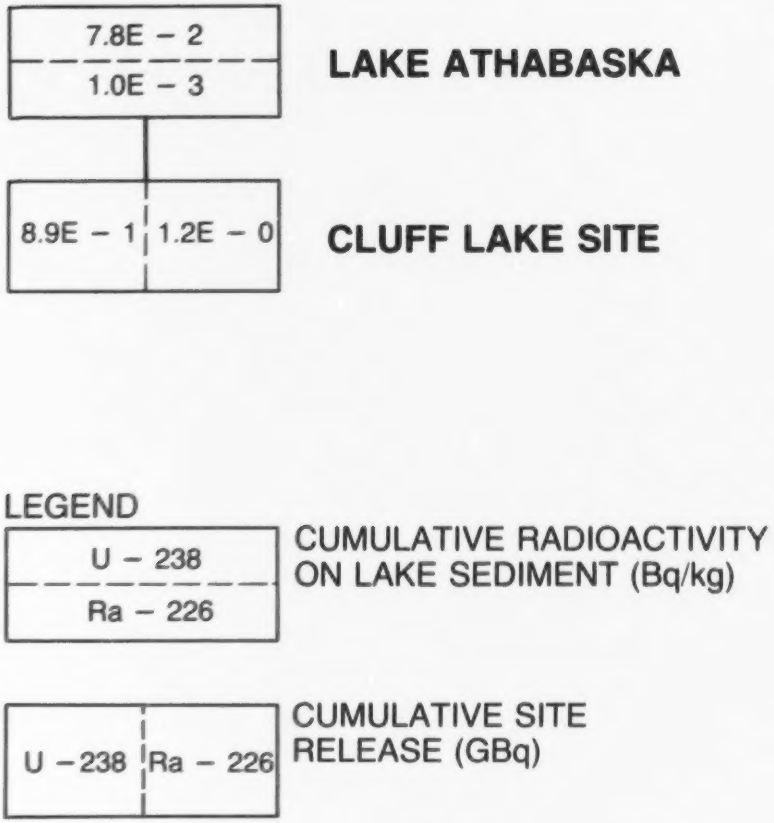
7.8E – 2
1.0E – 3
LAKE ATHABASKA
8.9E – 1
1.2E – 0
CLUFF LAKE SITE
LEGEND
U – 238
Ra – 226
CUMULATIVE RADIOACTIVITY ON LAKE SEDIMENT (Bq/kg)
U – 238
Ra – 226
CUMULATIVE SITE RELEASE (GBq)

FIGURE 9 RADIOACTIVITY DEPOSIT IN SEDIMENTS OF WATERSHEDS DRAINING INTO LAKE ATHABASKA

The annual dose equivalent was calculated by assessing the dose attributable to appropriate diet and lifestyle habits. For doses from aquatic sources, these are:

water consumption	800 L/a
wild rice consumption	1 kg/a
fish consumption	44 kg/a
time spent walking beach & sunning	92 h/a
time spent swimming & bathing	114 h/a

TABLE 18 AQUATIC PATHWAY - DOSES TO MOST HIGHLY EXPOSED INDIVIDUAL IN CRITICAL GROUP (μSv/a)

	Internal			External		
Nuclide	Rice	Fish	Water	Water	Beach	Total Dose (Radionuclide)
U-238	0.130E-3	0.258E-2	0.517E-2	0.135E-6	0.308E-5	0.815E-2
U-234	0.150E-3	0.331E-2	0.602E-2	0.865E-8	0.866E-7	0.948E-2
Th-230	0.340E-5	0.149E-3	0.905E-4	0.605E-10	0.677E-9	0.243E-3
Ra-226	0.406E-4	0.179E-2	0.650E-3	0.252E-6	0.878E-5	0.249E-2
Pb-210	0.571E-5	0.627E-2	0.114E-2	0.363E-9	0.602E-8	0.742E-2
Po-210	0.168E-7	0.923E-3	0.336E-3	0.450E-13	0.305E-10	0.128E-2
Total (Pathway)	0.347-3	0.153E-1	0.134E-1	0.396E-6	0.120E-4	0.291E-1

Fish consumption is deliberately set at a high level to avoid underestimating the annual dose rate to the most highly exposed individual.

Doses from Fish Ingestion

For fish, the annual dose is (fish consumption kg/a) x (radionuclide concentration in fish Bq/kg) x (dose conversion factor Sv/Bq ingested). A similar calculation is performed for dose from water consumption.

- Concentration of radionuclides in fish is calculated by multiplying the radionuclide concentrations in water by the appropriate bioconcentration factor for aquatic biota. The bioaccumulation factors listed in Table 19 are obtained.
- Radionuclide concentrations in water are determined by aquatic modelling. Results are given in Section 6.3 of this report.

External Exposure

a) Annual water immersion dose = (water concentration Bq/L) x (dose conversion factor Sv/a per Bq/L) x (percentage of year spent immersed)

TABLE 19 BIOACCUMULATION FACTORS - AQUATIC BIOTA
(Bq/g tissue per Bq/g water)

Radionuclide	Plankton	Invertebrates	Fish	Fish Eggs
Uranium	1 000	100	10	10
Th-230	1 500	500	150	100
Ra-226	2 500	1 000	100	100
Pb-210	200	100	275	10
Po-210	2 000	1 000	640	10

b) Annual beach walking dose = (percent of time spent beach walking) x (dose conversion factor Sv/a per Bq/kg) x (beach sediment contamination Sv/kg)

Calculations (a) and (b) are repeated for each radionuclide.

6.3 Collective Dose Commitment - Aquatic Impact

Annual collective doses for the Watershed Population are:

Internal	0.729E-1 person-Sv/a
External	0.572E-5 person-Sv/a
Total (Aquatic Sources)	0.729E-1 person-Sv/a

The collective dose consists of:

(i) dose received through the year from external sources; and

(ii) dose commitments from radionuclide uptakes received during the year.

Collective doses are listed in Table 20. For collective dose calculations, the Watershed Population includes:

(i) residents of the Wollaston Lake community;

(ii) all other residents of the Lake Athabasca and Reindeer Lake watersheds;

(iii) all individuals consuming fish exported from these watersheds.

Residents and transients in the Watershed Population are assumed to have lifestyles similar to Critical Group members, except that only 32% of them engage in swimming and 12% in other beach activities. People consuming exported fish are widely scattered and assumed to have no other contact with contaminated water.

TABLE 20 AQUATIC PATHWAY - COLLECTIVE DOSES TO THE WATERSHED POPULATION (person - Sv/a)

	Internal			External		
Nuclide	Rice	Fish	Water	Water	Beach	Total Dose (radionuclide)
U-238	0.518E-6	0.372E-5	0.210E-4	0.175E-8	0.145EW-8	0.252E-4
U-234	0.611E-6	0.432E-5	0.245E-4	0.113E-9	0.403E-10	0.294E-4
Th-230	0.676E-6	0.179E-6	0.180E-6	0.388E-13	0.154E-12	0.366E-6
Ra-226	0.190E-6	0.255E-5	0.303E-5	0.298E-9	0.363E-8	0.577E-5
Pb-210	0.181E-7	0.664E-5	0.359E-5	0.359E-12	0.240E-11	0.103E-4
Po-210	0.379E-7	0.100E-5	0.757E-6	0.316E-15	0.119E-12	0.180E-5
Total (Pathway)	0.138E-5	0.184E-4	0.531E-4	0.586E-9	0.512E-8	0.729E-4

6.4 Effects of Changes in Aquatic Source Terms

Calculations of radiological impact are based on the aquatic source terms presented in Section 2. While changes in the source terms could alter the radiological impact of a particular mine site, it should be emphasized that for the modelling methodology used in this study, there is a site-specific linear relationship between the magnitude of a radionuclide release and its subsequent impact. Therefore, impact calculations for each radionuclide at each mine site can be crudely adjusted to reflect changes in source terms.

Adjustments of this nature may be required at the Cluff Lake site where available data suggest an increasing trend in uranium loadings (Section 2). Assuming a threefold increase in the uranium source term, relative to that given in Table 3, uranium concentrations in Lake Athabasca water and sediment would be expected to increase by a factor of almost three times those given in Tables D.1 and D.2, Appendix D. Individual dose rates and population dose commitments would increase by the same factor. In this case, the adjustment is simplified by the fact that dose contributions attributable to uranium released from other mine sites are negligible as far downstream as Lake Athabasca. In general, however, all contributing sources should be considered in adjusting radiological impacts for source term changes.

7 DOSES TO BIOTA

7.1 Doses to Terrestrial Biota

Two dose components are calculated for terrestrial biota:

- internal doses from radionuclides incorporated within the organism; and
- external doses from both air and ground contamination.

Doses for each component vary between species, depending on the nature of the species. The estimated doses to terrestrial biota are summarized below.

Significance of Biota Doses. Available data indicates that no species will receive doses which are biologically significant. Radiological impacts on species are evaluated according to their effects on the species population (IAEA 1979; Bayda Commission 1978; other references as below).

In general, radiosensitivity of life forms is inversely proportional to their size; smaller animals can be expected to resist a given dose better than can the larger ones.

7.1.1 General Exposure of Terrestrial Biota.

General External Exposure. External dose rates in air have been calculated for Hidden Bay biota. The dose rates are:

Dose From Air Contamination (µGy/a)	Dose From Ground Contamination (µGy/a)	Location Total (µGy/a)
3.14	2.47	5.61

All mammals and plants are subject to exposures of this magnitude. Actual doses received in Grays from this source will depend upon the size and tissue make-up of the species.

Significance of General External Exposure. These exposures are expected to have no detectable effects. Plants exposed to dose rates in the tens to hundreds of milligrays range showed no adverse effects. Lethal dose rates are in the order of tens of Grays per year (Bayda Commission 1978).

Internal Exposure from Inhalation. Radon doses are expected to be more significant than doses from particulate inhalation. No dosimetry or lung metabolic data are readily available for moose, caribou, beaver, or other fauna of northern Saskatchewan. Inhalation doses are therefore not calculated.

7.1.2 Mammals.

Methodology Summary. Doses to muscles and organs are calculated according to the methods described in Section 4 using available data on radionuclide distribution in animals (Persson 1971 and 1972; IAEA 1979; Nelson and Rust 1967).

For gamma dose calculations, muscle geometries assumed were cylinders in:

caribou	20 cm radius x 140 cm long; $\bar{g}$ = 180 cm
moose	30 cm radius x 200 cm long; $\bar{g}$ = 220 cm
beaver	10 cm radius x 50 cm long; $\bar{g}$ = 106 cm

Table 21 shows the annual organ doses for moose, caribou, and beaver. The doses shown account for radionuclides ingested in the complete diet of the animal. Diet components considered are terrestrial forage, aquatic plants, and water.

The highest organ doses to caribou were:

muscle	0.510E-1 µGy/a
liver	0.672E0 µGy/a
kidney	0.826E0 µGy/a
bone	0.343E+1 µGy/a

Doses to moose and beaver organs are about 1% of those for caribou. Further details are given in Appendix C.

No quantitative estimates of biological effect were made for mammal doses.

7.1.3 Doses to Lichens.

Methodology Summary. Doses are calculated using MILDOS - derived air concentrations and air-lichen radionuclide distribution factors from the literature (UNSCEAR 1977; Persson 1971, 1972, 1973). For gamma dose calculations, lichens are modelled as spheres of 5 cm radius; $\bar{g}$ = 47 cm.

Doses to lichens are calculated because they are likely to receive the highest dose of any plant in the mining areas. Maximum estimated dose to lichens is 2.1 mGy per year. This is the internal dose from radionuclides taken up directly from the air, including deposited radionuclides. Table 22 shows dose contributions from each nuclide, for each of the three radiations considered. These doses do not include external dose contributions from the surrounding atmosphere or ground.

TABLE 21 ESTIMATED RADIATION DOSES TO MAMMALS FROM INGESTION OF FOOD AND WATER

Mammal (Habitat: Hidden Bay)	Radionuclides	Concentrations in Meat (Bq/kg)	Radiation Doses (µGy/a)			
			Muscle	Liver	Kidney	Bone
Caribou	Uranium*	0.356E-1	0.177E+1	0.530E+2	0.530E+2	0.177E+3
	Th-230	0.103E-1	0.245E+0	0.735E+1	0.735E+1	0.245E+2
	Ra-226	0.322E+0	0.701E+1	0.701E+1	0.701E+1	0.210E+4
	Pb-210	0.150E-1	0.430E+0	0.228E+2	0.989E+1	0.172E+3
	Po-210	1.58E+0	0.416E+2	0.582E+3	0.749E+3	0.749E+3
			0.510E+2	0.672E+3	0.826E+3	0.343E+4
Moose	Uranium*	0.908E-2	0.451E+0	0.135E+2	0.135E+2	0.451E+2
	Th-230	0.769E-4	0.183E-2	0.183E-2	0.549E-1	0.183E+0
	Ra-226	0.923E-3	0.219E-1	0.219E-1	0.219E-1	0.657E+1
	Pb-210	0.316E-3	0.841E-2	0.445E+0	0.193E+0	0.336E+1
	Po-210	0.718E-2	0.173E+0	0.242E+1	0.311E+1	0.311E+1
			0.660	0.164E+2	0.169E+2	0.583E+2
Beaver	Uranium*	0.436E-2	0.217E+0	0.651E+1	0.651E+1	0.217E+2
	Th-230	0.124E-4	0.298E-3	0.894E-2	0.894E-2	0.298E-1
	Ra-226	0.392E-3	0.937E-2	0.937E-2	0.937E-2	0.281E+1
	Pb-210	0.202E-4	0.576E-3	0.305E-1	0.132E-1	0.230E+0
	Po-210	0.217E-2	0.570E-2	0.798E-1	0.103E+0	0.103E+0
			0.232E+0	0.659E+1	0.659E+1	0.248E+2

* Includes U-238, Th-234, Pa-234, U-234.

No specific quantitative estimate was made of the biological effects of doses to lichens although lethal doses to algal cells which form a portion of the lichen symbiont are in the order of thousands of Grays.

TABLE 22 ESTIMATED RADIATION DOSES TO LICHENS AROUND THE RABBIT LAKE SITE (mGy/a)

	Alpha	Beta	Gamma	Total
Uranium*	0.134E+1	0.129E+0	0.157E+0	0.147E+1
Th-230	0.175E+0	-	-	
Ra-226	0.165E+0	-	-	
Pb-210	0.514E-1	0.454E-2	0.144E-1	0.559E-1
Po-210	0.180E+0	-	-	
Total	0.191E+1	0.134E+0	0.171E+0	0.204E+1

* Includes U-238, Th-234, Pa-234, U-234.

Diet of Mammals. Dietary assumptions are as follows:

	Consumption		
	Moose	Caribou	Beaver
Water Consumption	50 L/d	15 L/d	2 L/d
Terrestrial Forage	27 kg/d	7 kg/d (Lichen)	0.7 kg/d
Aquatic Plants	2 kg/d average	-	1 kg/d

Food quantities are fresh weight. Radionuclide concentration in the meat of each mammal is given by:

(Daily radionuclide intake) x (Feed-to-meat transfer factor)

Feed-to-meat transfer factors (Bq/kg meat per Bq/d intake) are listed in Table 23.

7.1.4 **Radionuclide Uptakes by Plants.** Table A.4 in Appendix A lists soil-to-plant radionuclide transfer factors for plant species considered in the diets of humans and animals. The factors are used to calculate plant radionuclide concentrations. These figures are used in calculating dietary uptakes by humans and animals. Plant doses are not calculated.

TABLE 23 TRANSFER FACTORS FROM FEED TO MEAT (days/kg)

	Caribou[2]	Beef[2]	Milk[2]	Poultry	Egg
Uranium[1]		3.4E-4	6.1E-4	4.0E-3	2.0E-3
Th-230		2.0E-4	5.0E-6	4.0E-3	2.0E-3
Ra-226	6.7E-3	5.1E-4	5.9E-4	5.0E-4	2.0E-5
Pb-210	1.1E-3	7.1E-4	1.2E-4	2.0E-3	2.0E-3
Po-210	3.3E-2	1.2E-2	1.4E-4	4.0E-4	1.8E-2

[1] Includes U-238, Th-234, Pa-234, U-234.

[2] Beef and milk values for U, Th, Ra and Pb from NUREG-0706, 1980; caribou values from McDowell-Boyer et al., 1979 and UNSCEAR, 1977; all other values from NUREG/CR-0553, 1979.

7.2 Doses to Aquatic Biota

Aquatic biota are all exposed to:

- internal radiation from ingested radionuclides;
- external radiation from radionuclides in water;
- external radiation from radionuclides in sediment.

Doses from each component to each species vary according to the size and habits of the species.

Because of the magnitude of radiation absorption coefficients for water, there are no general external radiation components which affect every organism.

Doses to the most exposed aquatic biota are shown in Table 24. In summary, the maximum estimated doses from incorporated radionuclides and radionuclides in the water are as follows:

fish	0.423E-4 mGy/d
fish eggs	0.356E-4 mGy/d
plankton	0.101E-2 mGy/d
invertebrates	0.378E-3 mGy/d

These are dose rates to biota in the water which receive the tailings liquid discharge from the Rabbit Lake mining/milling site. The biota originate in Otter Bay and migrate up the tailings creek as far as chemical concentrations permit. Doses are expressed in mGy per

TABLE 24 DOSE RATES TO AQUATIC BIOTA IN WATERS RECEIVING TAILINGS DISCHARGE - OTTER BAY (μGy/hr)

Radionuclide	Plankton	Crayfish	Clam	Fish	Fish Embryo
Uranium*					
Incorporated	0.410E-1	0.147E-1	0.147E-1	0.147E-2	0.137E-2
Water	0.104E-3	0.543E-7	0.543E-7	0.543E-7	0.543E-7
Th-230					
Incorporated	0.474E-4	0.525E-4	0.525E-4	0.158E-4	0.105E-4
Water	0.735E-7				
Ra-226					
Incorporated	0.651E-3	0.841E-3	0.841E-3	0.841E-4	0.841E-4
Water	0.607E-6	0.101E-8	0.101E-8	0.101E-8	0.101E-8
Pb-210 (+ Daughters)					
Incorporated	0.183E-4	0.321E-4	0.321E-4	0.883E-4	0.383E-6
Water	0.342E-6	0.107E-6	0.107E-6	0.107E-6	0.120E-6
Po-210					
Incorporated	0.952E-4	0.154E-3	0.154E-3	0.986E-4	0.154E-6
Water	0.111E-6				

* Includes U-238, Th-234, Pa-234, U-234

day because lifespan of plankton is about 20-50 days, and fish eggs have a gestation period of about 3 months.

7.2.1 Methodology Summary.

Incorporated Radionuclides. Using the general equations of Section 4.3.2 of this report, dose rates were calculated for incorporated radionuclides at concentrations of 1 Bq/g in the organism. This was done for each radionuclide according to the methodology of IAEA No. 172 (1976). For fish eggs, doses from beta and gamma radiation are based on other work (Woodhead 1970 - see Appendix D).

Radionuclides in Water. Again, IAEA No. 172 was used as a basis for these calculations. Except for plankton, there is no alpha radiation dose to any organisms from radionuclides in water.

Radionuclides in Sediment. Dose rates in water from contaminated sediment were derived using the work of Cross (1967) and Beck (1972). An infinite plane source is considered. The sediment bed is divided into planes to account for self-absorption for both beta and gamma radiations (Table 25).

TABLE 25 DOSE RATES FROM CONTAMINATED SEDIMENTS IN WATERS RECEIVING TAILINGS DISCHARGE - OTTER BAY (μGy/h)

Radionuclides	Beta	Gamma
Uranium*	0.81E-1	0.153E-2
Ra-226	-	0.112E-4
Short-lived Rn Daughters		
Long-lived Ra Daughters**	0.168E-3	-

* Includes U-238, Th-234, Pa-234, U-234.

** Includes Pb-210, Po-210.

Radionuclide Bioconcentration Factors. All internal dose calculations for aquatic biota are based on their internal radionuclide content. The values of radionuclide content used in all calculations in this report are based on bioconcentration factors. These factors relate concentration in the organism to concentration in the surrounding waters for each radionuclide. Bioconcentration factors used in this report are based on published literature and are listed in Table 19.

Fish. For internal gamma radiation doses, the muscle of fish is modelled as a cylinder, 10 cm radius x 50 cm long; $\bar{g} = 41$ cm. The fish receive no alpha or beta doses from radionuclides in the water but are estimated to receive 20% of the maximum dose from contaminated sediments.

Fish Eggs. Doses are calculated from the established methodologies of IAEA No. 172 (1976) and Woodhead (1970). The egg is modelled as a sphere of 2 mm diameter. The outer regions of this sphere are (1) the eggshell itself, 20 microns thick and (2) a 200-micron thick shell immediately inside the eggshell. The 200-micron shell is the area where the fish embryo develops, attached to the shell. Since fish eggs are laid on gravel beds or on weeds in weed beds, doses from contaminated sediment are negligible.

Plankton. Phytoplankton and zooplankton are considered, each assumed to be 50 microns in diameter. This size is not an infinite absorber for alpha radiation. Plankton similarly receive alpha radiation from water because of their small dimensions. Studies in which these species have been irradiated at 20 µGy/d and more showed no observable effects (Turner 1975, Blaylock 1978). The maximum estimated daily dose to plankton in this study is 0.101E-5 Grays per day. Thus, no effects on plankton are expected.

Crustacea. These are crayfish in northern lakes. For gamma radiation dose calculations, crayfish are modelled as a cylinder of 1 cm radius by 5 cm long; $\bar{g} = 10.6$ cm (Focht et al. 1965). Alpha and beta emissions from radionuclides in water deliver no dose to these creatures. They receive doses from contaminated sediment.

Molluscs. Fingernail clams are considered. For gamma dose rate calculations, they are modelled as a sphere of 0.5 cm radius; $\bar{g} = 4.7$ cm.

Fish Eggs. Considerable research has been done on the radiosensitivity of fish eggs. The maximum estimated fish egg dose of 0.107E-5 Grays over a hatching period of 30 days is estimated to have no detectable effect on the hatching period. It is less than 0.03% of the dose required to affect fish recruitment (Till, Kay and Trabalka 1976; MacGregor 1972; Blaylock and Trabalka 1978; Templeton 1975).

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APPENDIX A

ATMOSPHERIC MODELLING DEVELOPMENT

A.1 ENVIRONMENTAL TRANSPORT MODELS AND PARAMETERS

A.1.1 Objectives of Atmospheric Modelling. Radiological impacts from the following atmospheric exposure pathways and terrestrial food webs were assessed:

a) inhalation
b) external exposure
c) edible vegetation-to-man
d) lichens-to-caribou-to-man
e) forage, aquatic plants-to-moose, beaver-to-man

The computer code UDAD (Uranium Dispersion and Dosimetry) developed by the Argonne National Laboratory (UDAD, 1979) was used to develop the MILDOS model employed in this study:

a) wind erosion of tailings particles
b) airborne radionuclide concentrations
c) ground contamination and re-suspension
d) external doses from contaminated ground and air

As far as was possible, site-specific environmental data and parameters were used as input for running MILDOS. Where such data were not available, default values provided in MILDOS were used if judged to be appropriate. Contamination levels in biota were calculated manually as MILDOS was not designed to consider the food webs in northern Canada. Mammals such as moose, beaver, and muskrat have semi-aquatic habitats which are not modelled by MILDOS. A separate model had to be developed for the air-to-lichen-to-caribou-to-man food-chain. The internal dosimetry models and parameters used in MILDOS are based on ICRP 2 (1959) with the exception of the lung model which is adapted from the report of the ICRP Task Group on Lung Dynamics (ICRP 1966) and the dosimetry model for Ra-226 which is based on ICRP 10A (1971). In the present study, more up-to-date data (ICRP 30 1979) using the committed effective dose equivalent approach (ICRP 26 1977) with appropriate weighting factors for individual organs, and a regional lung dose concept (ICRP 32, 1981) were used for internal dosimetry calculations. In view of the fact that dosimetry models for terrestrial animals are not well developed, only a limited assessment of doses received by terrestrial biota was made in this study.

A.1.2 Atmospheric Dispersion. The atmospheric dispersion model used in MILDOS is the standard straight-line crosswind-integrated Gaussian plume model with allowance for plume rise, plume depletion by deposition and settling, radioactive decay of radon and ingrowth of its short-lived daughter products, and the atmospheric mixing boundary. The NUREG/CR-2011 description of the model has been used to provide the following methodology.

The ground level air concentration at a receptor which is downwind a distance x and crosswind a distance y for a pollutant i from a source j is given by:

$$\chi(x,y,i,j,s) = \frac{Q\,(i,j,y,s)}{(\sqrt{\Pi/2}\,)\,\sigma_z\;\bar{u}\,(\pi x/8)}\;\exp\left(-\frac{h^2}{2\sigma_z^2}\right) \qquad (A.1)$$

where $\chi(x,y,i,j,s)$ = ground level air concentration, $TBq{\cdot}m^{-3}$
x = downwind distance, m
y = crosswind distance, m
i = pollutant i
j = source j
s = particle size
$Q(i,j,y,s)$ = emission rate, $TBq{\cdot}s^{-1}$
σ_z = vertical standard deviation of plume concentration, m
$\bar{u}$ = average windspeed, $m{\cdot}s^{-1}$
h = effective height of plume centerline, m
$\pi x/8$ = sector width at distance x, m

The vertical dispersion, σ_z, as a function of downwind distance x is calculated using the following empirical expression (Briggs 1974; Gifford 1976).

$$\sigma_z = (ax)\,(1 + bx)^{c} \qquad (A.2)$$

Constants a, b, and c given in the MILDOS manual (NUREG/CR-2011) as a function of atmospheric stability. These are summarized in Table A.1. For distances less than 100 metres from source to receptor, the value of σ_z at 100 metres is used.

These constants are generally applicable for flat smooth terrain. The authors recognize this is not appropriate for northern Saskatchewan. The roughness parameter in the area is highly variable and small lakes dot the area around the mine/mill regions. When MILDOS is modified to incorporate a more appropriate plume rise model, the σ_z

TABLE A.1 STABILITY CLASS PARAMETERS FOR EQUATION A.2

Stability Class	Pasquill Type	a	b	c
1 Extremely unstable	A	0.20	0.0	1.0
2 Moderately unstable	B	0.12	0.0	1.0
3 Slightly unstable	C	0.08	0.0002	-0.5
4 Neutral	D	0.06	0.0015	-0.5
5 Moderately stable	E	0.03	0.0003	-1.0
6 Very stable	F	0.016	0.0003	-1.0

values from Hosker (1974) should also be incorporated. These values are dependent on the roughness length.

The effect of mixing layer height on dispersion is considered only for unstable and neutral conditions (Paquill A-D types), as stable conditions limit the plume dispersion in the vertical direction. Equation A.2 is used to calculate vertical dispersion out to a downwind distance x_L, at which $\sigma_z = 0.47L$, where L is the vertical distance from the ground to the base of the stable atmosphere layer (mixing layer height). At the base of the stable layer, the concentration of the radionuclide will be one-tenth of that at the plume centerline, for the distance x_L. At distances between x_L and $2x_L$, the concentration (for non-stable conditions) is determined by a linear interpolation between Equation A.1 and Equation A.3.

$$\chi(x,i,j) = \frac{Q(i,j)}{\frac{\pi x}{8} L\bar{u}} \qquad \text{(A.3)}$$

where L = mixing layer height.

The mixing layer height L varies greatly with the season, day to day, and also diurnally. An estimate of mixing height for a given location can be made from figures presented in Holsworth (1972). The annual average height for the mixing layer L can be estimated from:

$$\frac{1}{L} = \frac{1}{2}\left(\frac{1}{L_{AM}} + \frac{1}{L_{PM}}\right) \qquad \text{(A.4)}$$

where L_{AM} = mean annual morning mixing height, m
L_{PM} = mean annual afternoon mixing height, m

The average annual mixing layer is provided by the user. Annual average values of L in northern Saskatchewan are 450 m (northwest) and 325 m (northeast), (Portelli 1979). In the present study, a mean value of 400 m is used. For downwind distances greater than $2x_L$, Equation A.3 is used to calculate concentration.

The six wind speed categories have the following averages in miles per hour: 1.5, 5.5, 10.0, 15.5, 21.5, and 28.0

The effective plume height, h, takes into account plume rise due to effluent momentum from a stack or vent and also vertical movement due to particle settling. The rise due to momentum is based on the model of Holland (1953):

$$h_m = \frac{1.5\ VD}{u} \tag{A.5}$$

where V = effluent exit velocity, $m.s^{-1}$
D = inside diameter of stack, m
u = average windspeed, $m.s^{-1}$

See section A.2 for cautions concerning the Holland model.

The vertical settling is based on a "titled plume model" where the downward movement of the plume is given by:

$$h_v = \frac{xV_s}{u} \tag{A.6}$$

where V_s is the settling velocity and is calculated from:

$$V_s(m/s) = 3x10^{-5}\ \rho_s d^2 \tag{A.7}$$

where d = particle diameter, m
ρ_s = density of particle, g/cm^3

For settling velocities less than 0.01 metre per second, the vertical settling is ignored. The effective plume height is thus:

$$h = h_s + h_m - h_v - h_r \tag{A.8}$$

where h_s = stack height above mill center, m

h_r = elevation of receptor above mill center (negative if below mill center), m

At individual receptor locations the concentration is weighted by the distance from the midline of the sector. This is calculated by adjusting the source strength to be:

$$Q(i,j,y) = \frac{\left(\frac{\pi}{8}x-y\right)}{\frac{\pi}{8}x} Q(i,j,o) \tag{A.9}$$

Thus any receptor within 22.5 degrees (one sector width) in either direction from the centerline of the wind direction sector receives some concentration. All other receptors receive zero. Area sources are given an additional lateral dispersion by using a "virtual point source" method. The size of the area source is provided by the user in km^2 and is then converted to a square source of equivalent area. The distance to the "virtual point", taken upwind from the area source is calculated by:

$$x_{vp} = \frac{8}{\pi} S \tag{A.10}$$

where S = length of the side of an equivalent square source

This distance is added to the downwind distance when calculating the sector width. It is recommended by the NRC that area sources larger than 0.1 km^2 be partitioned into area sources of size less than or equal 0.1 km^2. This limits the virtual point source distance to 0.8 km or less.

For receptors that cannot "see" the entire area source, a correction factor is applied. This factor is the ratio of that portion of the square source area within the 22.5-degree sector located upwind from the receptor (assuming the receptor is on the centerline of the sector) to that of the total source area.

Multiplying the concentrations by the fractional joint frequency of occurrence of wind speed, wind direction, and atmospheric stability factors in the meteorology at the site, the annual average concentration at receptor k for pollutant i from all sources j is calculated from:

$$\chi(i,k) = \sum_j \sum_f f\chi(x,y,i,j) \qquad \text{(A.11)}$$

where f = fractional joint frequency of occurrence by wind speed, wind direction, and atmospheric stability

Triple joint frequency data f (i,j,k) for representative northern Saskatchewan sites (Uranium City, Cree Lake and a synthetic Collins Bay data set) were obtained by running the "STAR" computer program at the Atmospheric Environment Service in Downsview, Ontario (see also discussion in Appendix A.2).

Radioactive decay during transit downwind is allowed for by effectively reducing the source term Q:

$$Q'(x) = Q(x) \exp \frac{-\lambda_r X}{u} \qquad \text{(A.12)}$$

where λ_r is the radioactive decay constant (s^{-1}). The ingrowth of radon daughters during transit is computed using the standard Bateman equations and converted to Working Level (WL) values as a function of distance.

A.1.3 Ground Deposition and Resuspension. Particulate materials tend to settle out of the plume through gravitational settling if the particles are relatively large, or deposit by physical processes such as wind-induced impaction on surfaces if the particles are small. Particulate settling and deposition depletes the plume. It also directly results in ground and vegetation contamination. In MILDOS, plume depletion is allowed for by effectively reducing the source term Q in Equation A.1 by a factor which is a function of the downwind distance, wind speed, release height, z, and the deposition velocity, V_d, defined as follows:

$$V_d = \frac{\text{Deposition Rate D } (\text{Bq/m}^2\cdot\text{s}^{-1})}{\text{Air Particulate Concentration } (\text{Bq/m}^3)} \qquad \text{(A.13)}$$

In MILDOS, the gravitational settling velocity V_s is first calculated, based on particle size and density. If V_s is greater than 0.01 m/s, then V_d is set equal to V_s. If V_s is less than 0.01 m/s, V_d is taken to be constant at 0.01 m/s. Typical values used in MILDOS are given in Table A.2

A particle density of 8.9 g/cm^3 applies to yellowcake, whereas the value of 2.4 g/cm^3 apples to tailings and soil particles. For Pb-210 and Po-210 generated in the air by airborne radon, the value of V_d used is 0.003 m/s.

TABLE A.2 DEPOSITION OF VELOCITIES USED IN MILDOS

Particle Size (μm)	Particle Density (gm/cm^3)	v_d (m/s)
1	8.9	0.01
1	2.4	0.01
5	2.4	0.01
35	2.4	0.088

As defined in Equation A.13, the ground deposition rate is V_d. The rate of loss of deposited material on the ground is $(\lambda_r + \lambda_w)\ C_g$,

where λ_r = radiological decay constant (s^{-1})
λ_w = environmental weathering loss constant (s^{-1})
C_g = ground surface concentration (Bq/m^2)

Then C_g as a function of deposition time t is given by:

$$C_g = \frac{(x)\ V_d}{\lambda_r + \lambda_w}\ 1\text{-exp}\left[-(\lambda_r + \lambda_w) / t\right] \qquad \text{(A.14)}$$

MILDOS uses a λ_w value of $4.4 \times 10^{-10}\ s^{-1}$, corresponding to a weathering loss half-life of 50 years. The value of C_g is computed for the end of a mine's operation lifetime, i.e., t is set equal to the projected operating time of the mine. UDAD also allows for the ingrowth of Pb-210 from Ra-226 deposited on the ground, and the ingrowth of Po-210 from Pb-210.

Resuspension is not treated as a loss mechanism in the calculation of ground concentrations, and therefore ground concentrations are not assumed to be increased by the deposition of the resuspended activity.

Particulate material deposited on the ground may re-enter the atmosphere by resuspension. The total air concentration is therefore a sum of the direct concentration as calculated in Equation A.11 and the concentration from resuspension. Under steady state conditions, the resuspended concentration (Bq/m^3) is given by:

$$X_r = K.C_g \qquad \text{(A.15)}$$

where C_g = uniform ground contamination (Bq/m^2)

K = resuspension coefficient (m^{-1}).

The value of K is not constant but decreases with time, a reflection of the observation that as the deposited material ages, it becomes more difficult to resuspend.

For small particles (< 10 μm), the initial value of K used in MILDOS is 10^{-5} m^{-1}. This value decreases with a weathering half-life of 50 days until a value of 10^{-9} m^{-1} is reached. This is then considered the terminal value. The value of K is assumed to be inversely proportional to the deposition velocity V_d.

The total air concentration is a sum of the direct air concentration calculated by Equation A.11 and the resuspended air concentrations given by Equation A.15, i.e.,

$$\chi(\text{Total}) = \chi + \chi_r \tag{A.16}$$

A.1.4 Transfer to Vegetation and Forage. Contamination in vegetation and forage arises both from direct aerial deposition and from the uptake of radioactive materials deposited on soil. The concentration of radioactive material in vegetation is given by (NUREG-0706 1980):

$$C_v = \text{Foliar deposition} + \text{soil uptake}$$

$$= \frac{D f_v E_v (1 - e^{-\lambda v t})}{\lambda_v Y_v} + \frac{C_g B_v}{\rho} \tag{A.17}$$

where C_v = concentration in vegetation (Bq/kg)
D = deposition rate (Bq/m^2-s)
f_v = fraction of total deposition retained on vegetation surface
E_v = fraction of foliar deposition reaching edible parts of vegetation
λ_v = effective removal constant; sum of weathering loss rate constant and radioactive decay constant (s^{-1})
Y_v = vegetation yield (kg/m^2)
t = period of time over which foliar deposition occurs (s)
C_g = surface ground concentration (Bq/m^2) as given by Equation A.14
ρ = effective surface soil density (kg/m^2)
B_v = transfer factor from soil to plant (Bq/kg plant per Bq/kg soil)

The deposition rate D is given by:

$$D = \chi(\text{total}).\ V_d \tag{A.18}$$

where χ(total) = the total air concentration (Bq/m^3) as given by Equation A.16
V_d = the deposition velocity (m/s)

Values of V_d are given in Table A.2. Choice of parameter values in Equation A.17 is discussed below:

a) Vegetation Yield

Hoffman et al. (1979) reviewed and carried out a statistical analysis of available data. These are shown in Table A.3.

TABLE A.3 VEGETATION YIELD DATA (Hoffman et al., 1979)

	Vegetation Yield (kg/m^2)	
Vegetation	Median Value	Range
Forage Grasses (dry wt.)	0.33	0.04 - 1.6
Forage Grasses (fresh wt.)*	1.1	0.13 - 5.3
Leafy Vegetables (fresh wt.)	2.1	0.36 - 5.3
Non-leafy Vegetables (fresh wt.)	0.65	0.17 - 1.6
Below-ground Vegetables (fresh wt.)	2.8	0.79 - 5.6

* Assume 70% moisture content.

The annual production of grasses and herbs in northern Saskatchewan is given as 0.0207 kg/m^2 (dry wt) for a young stand (1-10 years). The cumulative yield for an established stand is assumed to correspond to 5 years' growth, or about 0.1 kg/m^2 (dry wt). Assuming the moisture content to be about 70 percent, the yield on a fresh weight basis would be about 0.3 kg/m^2. This value lies toward the low end of the range given by Hoffman et al. (1979) and reflects the generally lower productivity in northern Saskatchewan.

Annual new growth of the edible portions of shrubs (eaten by browsing animals such as moose) is given as 0.0107 kg/m^2 (dry wt) for a young stand of 1-10 years, increasing to about 0.05 kg/m^2 (dry wt) for an older stand. It seems reasonable to assume for an established stand and a cumulative yield of 0.1 kg/m^2 (dry wt), i.e., the same as that for grasses and herbs.

Site-specific yield data on edible vegetables for human consumption were not available. Allowing for the lower productivity in northern Saskatchewan, vegetable yields yields were assumed to be about a factor of 2 lower than the median values in Table A.3.

In summary, vegetation yields used in the present study are:

forage grasses, herbs, shrubs	- 0.1 kg/m^2 (dry wt)
	- 0.3 kg/m^2 (fresh wt)
above ground edible vegetables	- 1.0 kg/m^2 (fresh wt)
below ground edible vegetables	- 1.5 kg/m^2 (fresh wt)

b) Foliar Retention Fraction

Chamberlain (1970) has shown that for forage grasses, f_v is related to the vegetation yield (Y_v) as follows:

$$f_v = 1 \exp (\mu Y_v) \tag{A.19}$$

where the constant μ was found to range between 2.3 and 3.3 m^2/kg (dry wt) when Y_v has units kg/m^2 (dry wt). Miller (1976a) reviewed available data and concluded that the above relationship would give a reasonable estimate of f_v for forage grasses but is in general unsatisfactory for edible vegetation. Based on a value of 0.1 kg/m^2 (dry wt) for Y_v and 2.8 m^2/kg for μ, f_v was calculated according to Equation A.19. The resulting value of 0.25 was used in the present study. The ratio $f_v/Y_v = 0.25/0.1 = 2.5$ m^2/kg (dry wt) agrees well with the survey of Hoffman et al. (1979) who found f_v/Y_v values for forage grasses ranging from 1 to 4 m^2/kg (dry wt) and a median value of 1.8 m^2/kg (dry wt).

Reliable f_v values for edible vegetables were not available. A value of 0.2 was recommended in U.S. NRC Regulatory Guide 1.109 (1977) for deposition of particulates. Experiments (Witherspoon and Taylor 1971) using 1 to 44 μm diameter particles showed f_v values of about 0.4 for soybean and sorghum plants, with values for deposition on the leafy parts of these plants ranging from 0.09 to 0.32. For the present study, the value used for forage grasses, i.e., 0.25, was also used for edible vegetation.

c) Effective Removal Constant

The effective removal constant λ_v is related to the weathering loss half-life T_w by $\lambda_v = 0.693/T_w$ (0.693 - ln 2). Values of T_w reviewed by Hoffman et al. (1979) for grasses and other vegetation showed a range of 8.7 to 28 days, with a median value of 16 days. This median value agrees very well with the value of 14 days used in the U.S. NRC Regulatory Guide 1.109 (1977). In the present study, the value 14 days was used. The corresponding λ_v value is 0.0495 d^{-1}, or 5.7×10^{-7} s^{-1}.

d) Fraction of Foliar Deposition Reaching Edible Parts of Vegetation

E_v is assumed to be 1 for above ground vegetation and 0.1 for below ground vegetation (NUREG-CR-2011 1981).

e) Effective Surface Soil Density

ρ is taken to be 150 kg/m^2, corresponding to a soil depth of 10 cm. This relatively low value takes into account the high organic content of the boreal forest soil.

f) Deposition Time

For edible vegetables harvested annually, the deposition time is essentially the duration of the growth season, which is assumed to be about 60 days in northern Saskatchewan. For forage grasses and herbs, the deposition time is long compared to the weathering loss half-life so that the term $(1-e^{-\lambda_v t})$ in Equation A.17 may be taken to be 1.

g) Soil-to-Plant Transfer Factors

Soil-to-plant transfer factors from NUREG-0706 (1980) and Moffet and Tellier (1977) were used for modeling purposes (Table A.4). Other transfer factors from McDowell-Boyer et al. (1979) are given in Tables A.5a and A.5b to indicate the range of variability among crop plants.

A.1.5 Transfer to Lichens. Lichens are a major link in the air-to-lichens-to-caribou-to-man food-chain in northern Saskatchewan. Because they do not have a root system, they derive their nutrition preferentially from the air. They present high sorption area, live for many years, and are very efficient accumulators of practically all of the fallout radionuclides, both natural and artificial. Because soil uptake is negligible, direct radionuclide transfer factors between air and lichens are probably the best to use for impact assessments, where such data are available.

The behaviour of the natural fallout radionuclide Pb-210 (from decay of atmospheric Rn-222) in the air-to-lichens-to-caribou-to-man food-chain has been very extensively studied. Lichen concentrations are given in UNSCEAR (1977).

Average surface air Pb-210 concentrations are given in UNSCEAR (1977) as 0.51 Bq/m^3. The average value for Barrow, Alaska, over a 12-month period in 1975-1976 was 0.55 Bq/m^3 and the average value for Moosonee, Ontario in 1975 was 0.51 Bq/m^3 (EML-349 1979). Values given in NCRP-45 (1975) for continental North America range

TABLE A.4 SOIL-TO-PLANT TRANSFER FACTORS[1]

	Above Ground Vegetation	Below Ground Vegetation	Forage	Pasture Grass
Uranium[2]	2.5E-3	2.5E-3	2.5E-3	2.5E-3
Th-230	4.2E-3	4.2E-3	4.2E-3	4.2E-3
Ra-226	1.4E-2	3.0E-3	8.2E-2	1.8E-2
Pb-210	4.0E-3	4.0E-3	3.6E-2	2.8E-2
Po-210[3]	1.0E-3	1.0E-3	1.0E-3	1.0E-3

[1] Units are in $\frac{\text{Bq/kg plant}}{\text{Bq/kg soil}}$

[2] Includes U-238, Th-234, Pa-234, U-234.

[3] Factors for Po from Moffett and Tellier (1977); all other factors from NUREG-0706 1980.

from 0.29 to 0.95 Bq/m^3. Using a mean concentration of 0.51 Bq/m^3 and the mean lichen concentrations from UNSCEAR (1977), transfer factors for Pb-210 from air to lichen are:

$$\frac{275{,}000 \text{ Bq/kg}}{0.51 \text{ Bq/m}^3} = 5.4 \times 10^5 \text{ m}^3\text{/kg (dry wt)}$$

$$\frac{109{,}900 \text{ Bq/kg}}{0.51 \text{ Bq/m}^3} = 2.1 \times 10^5 \text{ m}^3\text{/kg (fresh wt)}$$

Po-210 concentrations in lichens are approximately equal to that of the parent Pb-210 (UNSCEAR 1977). A separate transfer factor for Po-210 is not necessary.

Reliable direct transfer data from air to lichens for other radionuclides were not available. For these radionuclides, a deposition model was used. The cumulative concentration in lichens C_v (Bq/kg) is given by:

$$C_v = \frac{fv\, D}{\lambda_v Y_v S}(1-e^{-\lambda_v t}) \qquad \text{(A.20)}$$

where S is a snow cover factor that allows for the effective reduction in deposition on lichens during the winter months. All other symbols have the same meanings as used in Equation A.17.

The deposition rate D is given by $D = \chi \cdot V_d$ as defined in Equation A.18. The deposition velocity (m/s) is v_d. MILDOS uses a default v_d value of 0.01 m/s for

TABLE A.5a SOIL-TO-PLANT TRANSFER FACTORS FOR Pb-210*

Edible Plant Portion	Mean Transfer Factors ($x10^3$)	Number of Derived Values	Values or Ranges ($x10^3$)
Vegetables (fresh weight)			
Bean	7.8	1	7.8
Broccoli	0.8	2	0.6 - 1.0
Cabbage	4.8	1	4.8
Carrot	15.0	3	2.0 - 24.0
Cauliflower	0.5	2	0.4 - 0.5
Corn	4.2	1	4.2
Lettuce	2.9	15	0.5 - 9.9
Potato	4.2	1	4.2
Radish	3.6	4	0.3 - 7.1
Spinach	2.9	1	
Tomato	2.5	1	2.5
Unweighted Average	4.5	33	0.3 - 24.0
Forage Grasses and Herbs (dry weight)			
Oat tops	90	12	20 - 290
Fodder	40	1	40
Forage	140	2	90 - 190
Unweighted Average	90	15	20 - 290

* Transfer factor is defined as $(\text{Pb-210})_{\text{plant}} / (\text{Pb-210})_{\text{soil}}$.

particulates. This mean value is likely too high except at locations fairly close to the source where the particulate sizes may be relatively large. At larger distances, the plume will be depleted of the larger particulate materials, and a v_d value of 0.002 m/s recommended for atmospheric aerosols (Travis et al., 1979) may be more appropriate. The fractional retention factor f_v is assumed to be 0.95 (Garner 1972). The snow cover factor S is assumed to be 2.

The standing crop biomass Y_v of terrestrial lichens in northern Manitoba (mainly Cladonia alpestris) was measured by Miller (1976b). The mean value from 31 locations was 0.46 kg/m^2 fresh weight, range 0.056 to 0.68 kg/m^2.

TABLE A.5b SOIL-TO-PLANT TRANSFER FACTORS FOR Ra-226*

Edible Plant Portion	Mean Transfer Factors (x10^2)	Number of Derived Values	Range (x10^2)
Vegetables (fresh weight)			
Beet	1.8	2	0.6 - 3.0
Cabbage	1.6	5	0.01 - 4.0
Carrot	2.0	2	1.0 - 3.0
Potato	0.3	3	0.07 - 0.6
Unspecified	0.1	1	0.1
Unweighted Average	1.2	13	0.01 - 4.0
Forage Grasses and Herbs (dry weight)			
Clover	21.	6	1.1 - 4.8
Fescue	2.8	1	2.8
Grass	13.	28	2.0 - 63
Herbage	0.8	1	0.8
Rye Grass	24.	24	2.0 - 62
Unweighted Average	12	60	0.8 - 63

* From review by McDowell-Boyer et al., 1979).

The retention half life of material deposited on lichens is very long. Values reported in the literature are:

- Cs-137 - 2.7 to 17 years, mean 8 years (Mattsson 1975)
- Sr-90 - 1.5 years (Persson 1971)
- Pb-210 - 7 years (UNSCEAR 1977)
- Pu-239 - 6.1 years (Holm and Persson 1975)

The effective retention half life is reduced by caribou grazing. Grazing half lives of 5 years (Garner 1972) and 6 to 7 years (Mattsson 1975) have been reported. For the present study, an average retention half life of 6 years was used. The corresponding λ_v value is $0.693/6 = 0.116$ year^{-1}.

Using a v_d value of 0.003 m/s, a deposition period t of 15 years, and parameter values discussed above, it can be derived from Equation A.20 that the air-to-lichen transfer factor is about 7×10^5 m^3/kg for Pb-210 and provides some measure of confidence in the model and parameter values used.

A.1.6 Transfer from Feed-to-Meat. The concentration factor F_m from animal feed-to-meat is usually expressed as the ratio of meat concentration to animal feed rate rather than as the ratio of meat-to-feed concentrations since the animal's diet tends to be varied and include different components having different radionuclide concentrations. It has units of Bq/kg per Bq/d intake, or d/kg. If the mean concentration in the feed is known, F_m may be related to the meat-to-feed concentration ratio as follows:

$$F_m \text{ (d/kg)} = \frac{\text{Meat Concentration (Bq/kg)}}{\text{Feed Concentrations (Bq/kg)}} \div \text{Feed Rate (kg/d)} \qquad \text{(A.21)}$$

Meat-to-feed concentration ratios of Pb-210, Po-210, and Ra-226 are derived from studies of natural fallout levels in caribou in the air-to-lichens-to-caribou-to-man food-chain, as well as experimental data obtained on other mammals such as cows, sheep, and swine. For U and Th, concentration ratios are derived from F_m data given for beef (NUREG-0706 1980) using a feed rate for beef cattle of 12 kg dry wt/day (Ng et al., 1979). These data are summarized in Table A.6 and are assumed to apply to land mammals. Concentration factors (F_m) (Table 23 in main report) for caribou, moose, and beaver as representative mammals are calculated from Equation A.21 using the following diet intake data:

- caribou 2.8 kg dry wt/day (Garner 1972)
- moose 4.8 kg dry wt/day (Velovsky et al., 1975)
- beaver 0.4 kg dry wwt/day (estimated from Banfield 1974)

Extrapolation of meat-to-feed concentration ratio data obtained for large mammals to small mammals such as beaver is likely to be conservative (see discussion in Appendix C).

TABLE A.6 TRANSFER FACTORS FROM FEED-TO-MEAT (days/kg)

	Caribou[2]	Beef[2]	Milk[2]	Poultry	Egg
Uranium[1]		3.4E-4	6.1E-4	4.0E-3	2.0E-3
Th-230		2.0E-4	5.0E-6	4.0E-3	2.0E-3
Ra-226	6.7E-3	5.1E-4	5.9E-4	5.0E-4	2.0E-5
Pb-210	1.1E-3	7.1E-4	1.2E-4	2.0E-3	2.0E-3
Po-210	3.3E-2	1.2E-2	1.4E-4	4.0E-4	1.8E-2

1 Includes U-238, Th-234, Pa-234, U-234.

2 Beef and milk values for U, Th, Ra and Pb from NUREG-0706, 1980; caribou values from MCDowel-Boyer et al., (1979) and UNSCEAR (1977); all other values from NUREG/CR-0553 (1979).

A.2 ATMOSPHERIC EMISSION SOURCE TERM

This section provides information supporting the atmospheric Emission Source Terms given in Section 2 of the main repot.

A.2.1 Emissions - Mining Operations. Two types of mining operation can be used for uranium extraction in northern Saskatchewan. Underground mining will be used for parts of the operation at Cluff Lake; open pit mining methods are currently used at all three sites. Both approaches cause increased emissions of radon gas and radioactive particulate matter compared to natural levels escaping from undisturbed ore bodies since ore removal creates dust and increases radon releases.

For open pit mines, operations are sheltered from direct wind flows, and an area of recirculation may be created in the pit. Tests undertaken by Canada Wide Mines for the Midwest Project included wind tunnel assessments of this zone or recirculation in the pit. It was determined that the important parameters governing this recirculation are:

- wind speed
- orientation with respect to the wind
- depth of the pit

It was concluded that much of the particulate material released in a pit deeper than 30 m would not escape (Jandali 1980). Aerial observations of iron ore mining operations confirm this data. Winter flights over Schefferville, Labrador, and Antikokan, Ontario, show the red dust to be confined to the pit and roadway areas. For these reasons, the dust generated by open pit mining activities was ignored in this study. The Key Lake EIS (KLMC 1979) shows these values to be small when compared to fugitive emissions from ore storage and handling, a further reason to ignore this source contribution.

Radon Emission. This is a function of:

- radium concentration
- surface area exposed
- emanation coefficient
- diffusion properties of material

Generally, radon emision is defined in terms of a flux rate. Such a term is normalized on the basis of the amount of U_3O_8 present in the surface material. However, it can be affected by moisture content and, in northern Saskatchewan, by the presence of frozen ground or snow cover. In the absence of the site-specific data, a calculation of radon emission rates was based on normal summer conditions following the procedure outlined in NUREG/CR-0628. This requires that the following factors be defined:

- average grade of mineralization
- area exposed
- flux rate

In order to define these terms, it is necessary to make some assumptions. Among those made for this study are:

- ore and sub-ore have a constant lateral distribution,
- sub-ore grade is one half the mill cutoff grade,
- overburden mineralization is as defined in baseline documents, and
- the ore and sub-ore are approximately equal in the mining zone.

Radon flux rates are subject to great fluctuations. NUREG/CR-0628 reviews literature data. The values cited range from (2.3 - 10) x 10^{-3} Bq/cm^2·s/%U_3O_8. Data in the KLMC EIS report a radon flux rate of 0.034 Bq/cm^2·s/%U_3O_8 for an ore grade of 2.4% U_3O_8. Utilizing this flux rate factor, and on the basis of measured radon fluxes of 3.7 x 10^{-7} Bq/cm^2·s, the calculated overburden mineralization values would be 0.14 ppm U_3O_8. This would be expected since the overburden is outwash sand and has little mineralization. AMOK (1982) report radon flux rates from 8.5 x 10^6 to 1.5 x 10^8 atoms/m^2·s for uranium contents of 0.39 to 6.45 % U_3O_8. This translates to an average radon flux rate of 4.8 x 10^{-3} Bq/cm^2·s/%U_3O_8. This number can be considered in light of the moisture content in the underground mine area for which the calculation was made. High moisture contents will reduce the radon flux rate. Utilizing the data of Haywood et al cited in NUREG 0706, the radon flux from tailings can vary by 3.4 times as the tailings go from wet to dry. This would correspond to (4.8 - 20) x 10^{-3} Bq/cm^2·s/%U_3O_8. A value of 0.1 Bq/cm^2·s/%U_3O_8 was utilized for this study as an average of two reported values.

Radon generated from mining operations in this study was calculated on the above basis. Sources considered include:

- pit
- material handling above the pit rim
- discharge of dewatering water
- storage areas

The specifics for each mine are outlined in the following sections.

Fugitive Dust Emissions. Similar sources are considered for fugitive dust emissions. For the purposes of this study, equations from the work of Cowherd et al (1978) were selected to define the source terms. These are shown in Table A.7.

TABLE A.7 FUGITIVE EMISSION EQUATIONS

MATERIAL HANDLING EMISSIONS

$$Q = 0.0008\ (S/5)\ (U/8.5)/(M/2)^2\ (Y/4.6)$$

where Q = kg emitted per tonne (t) of ore handled
S = silt content of ore percent (5)
U = mean wind speed (km/h)
M = moisture content percent (12)
Y = dumping capacity (m^3) per vehicle

OPEN SOURCE WIND EROSION

$$Q = 0.38\ (e/0.184)\ (S/15)\ (f/25)/((P-E)/50)^2$$

where Q = source strength ($kg/m^2 \cdot a$) of exposed area
e = soil erodability $t/km^2 \cdot a$
S = silt content, percent
f = % time wind exceeds 19 km/h
$P-E$ = Thornthwaite's index.

STORAGE PILE WIND EROSION

$$Q = 0.2\ (S/1.5)\ (d/235)\ (f/15)\ (D/90)$$

where Q = emission rate in kg/t ore
d = number of dry days/a
D = total number of days the ore is in storage
f = percent time wind exceeds 19 km/h
S = silt content.

When attempting to predict the amount of dust that is generated by haulage, handling, and maintenance operations, it is worthwhile to remember that the visible dust that is emitted when material is being handled is usually of a large size range. Test data from Bennett et al. (1980) indicate that 77 percent of the particles released by loading operations in the sand and gravel industry exceed 30 µm in size. This material is subject to gravitational forces and settles out within 200 to 500 m of the source. The balance of the material, that which is finer than 30 µm, is transported by the wind. This material can be transported considerable distances downwind. Other test data can be cited to confirm this information. The equations used in this study predict the amount of material smaller than 30 µm that is released by the various operations.

While it is possible to estimate the fugitive source term for vehicular traffic, this was only done for Key Lake, and the release of radionuclide material is low (0.003 TBq/a). It is necessary to specify the mineralization of the roadbed material in each case to calculate these emissions. In addition, the size of the vehicles, their speed and configuration, and the number of wheels must be determined, along with the mileage travelled at each speed and for each weight class. Given the minor nature of this source at Key Lake, it was assumed that roadbed-vehicular emissions at other sites would be equally insignificant. The only exception to not providing estimates for haulage emissions is ore loss from trucks travelling between Collins Bay and Rabbit Lake. The method of determining this value is outlined later.

Quantification of Dust Terms. The terms common to the equations in Table A.7 must be defined. In addition, the sequence of operations should be determined. For all mines it was assumed that mining operations move the ore to the storage pile and the waste into storage. Wind erosion of the piles was attributed to the mining phase. Rehandling of the ore was considered a milling operation but it was included in mining terms for ease of modelling. The terms in the equations are defined as follows:

- % time that wind is greater than 19 km/h, frequency
  Key Lake 15.8 AMOK 16.2 Eldor 15.2 (from AES 1982)
- mean wind speed m/s
  Key Lake 3.3 AMOK 2.5 Eldor 3.5
- Thornthwaites index P.E. - 45 (NRCC 1982)
- average number of dry days 250 (AES 1982)

The values calculated by the equations refer to the total amount of material that is 30 µm in size. This is the traditional cutoff for the high volume sampling methods which were used to develop the data. Some larger material will move within a short distance of the source but it has been neglected at this time. Larger material has a faster settling velocity and is confined on site.

Factors such as freeze-thaw cycles and drying of material in storage cause weathering effects. These can be related to the duration of time in storage and assessed using the third equation in Table A.7.

No allowance for frozen ground, snow cover, or other mitigating factors has been included in the calculation of fugitive emissions. Generally, northern Saskatchewan experiences 4-6 months per year of frozen, snow-covered conditions. The continual action of ore and waste removal from the pit will result in continual exposure of surfaces, thereby limiting the mitigation effect. In the absence of better data, no allowance has been made for the attenuation. The result is that all fugitive values in this study may err on the high side by 15-25%.

A.2.1.1 Key Lake Atmospheric Emission - Mining.

Radon. Mining operations at Key Lake will consist of two open pit operations approximately 2 km x 1 km in extent. Both pits will operate together for a short period of time. Radon emissions have been postulated at the point where the G IV development is finished (1988). At that time, the first three expansions of the Deilmann pit are scheduled to be completed and the following elements should have been created:

- cobble ore pile (ultimate) 235 000 BCM* 670 m x 170 m x 12 m H
- special waste pile (ultimate) 400 000 BCM 360 m x 160 m x 12 m H
- waste area G-pit (ultimate) 2 @ 35 x 10^6 BCM 1350 m x 1150 m x 35 m H
  1 @ 1058 m x 1058 m x 45 m H
- Gaertner Pit (actual) surface 300 m x 1000 m; bottom 800 m x 100 m; depth 60 m; (side slopes 1:2)
- Deilmann Pit (actual) surface 1400 m x 530 m bottom 930 m x 230 m; depth 50 m; (side slopes 1:3)
- Ore storage (active) 28 000 LCM* triangular 280 m L x 10 m H x 20 m W.

Pit size is estimated from the amount of material removed: waste, special waste, ore, and cobble ore. Only 2 ha (2 x 10^4 m^2) of ore will be exposed at any time. This reduces the emission of radon from the ore. Radon emissions from the base of the pits can be calculated on the basis of overburden values and minimal ore exposed. This overestimates the release rate since the ore will be mixed with special waste and waste. The bottom 30 m are composed of waste rock assumed to have 0.015% mineralization. For the Dielmann pit, no ore is exposed at the time selected for calculation. The radon generation value was calculated from the radio of overburden to rock removed from the pit. Total rock removed will be 3 times the sand removed.

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* BCM = Bank cubic metres
LCM = Loose cubic metres

The areas of the different sources were calculated as shown in Table 4 in the main report. For calculation purposes, the cutoffs for U_3O_8 concentrations were:

- 1.4×10^{-5} % for overburden
- 2.5 % for ore
- 0.04 % for special waste
- 1/2 the mill cutoff of 0.03% for waste
- 0.5% for cobble ore

Material handling calculations were based on:

- mineralization (% U_3O_8)
- production rate (m^3)
- emanation factor (the amount of radon that escapes the mineral matrix)
- unit activity (TBq/m^3 % U_3O_8)

Conversion of tonnes of production to cubic metres utilized a density of 1.8 t/m^3. The emanation factor from KLMC data is 0.3. The unit radium activity was selected at 185×10^6 Bq/m^3-% U_3O_8. Production rates were estimated at:

- waste - 4.1×10^6 m^3/a
- ore - 1.2×10^5 m^3/a
- special waste - 1.8×10^4 m^3/a

Ore is handled 4 times before it enters the mill circuit at:

- pit to crusher
- crusher to storage
- storage to grinding
- grinding to mill

In the last stage, the material is handled as a slurry in a pipeline and no contribution to handling emissions was considered. Material handling emissions of radon are presented in Table 4 of the main report.

Radon gas will be released when water is discharged from the mine areas. On the basis of estimated dewatering volumes and a concentration of 185 Bq/L radium-226, mine water discharge will result in the release of 0.3 TBq/a of radon gas.

The mining of ore will result in an increase in radon emissions over the base case. The area covered by the various sources must thus be taken out of the regional production figure. With the low mineralization of the overburden, this results in a net reduction of less than 0.02 TBq/a.

As work progresses at the site, the Gaertner deposit will be mined out and backflooded. The cobble ore pile will be blended into the mill feed and thus reduce in size. The special waste and waste piles will grow. Exposed ore in Dielmann will replace that in the Gaertner pit. Dewatering of Gaertner will cease. A net reduction in radon emissions will therefore result.

Dust Releases

Haulage - Road Surfaces. As detailed in the introduction, emissions from haulage operations were estimated at the Key Lake site. Total emission from road surfaces were calculated at 631 t/a of material 30 μm and smaller in size. This would yield a 0.07 Ci/a release of Uranium-238, based on 1.5×10^{-4} g U_3O_8/g dust which is the value of the waste cutoff (0.015 % U_3O_8). If the value were closer to overburden levels (1.4×10^{-5} % U_3O_8), the total release would be 26×10^6 Bq/a. These values assume no fine particle enhancement factors for the amount of radioactivity. The calculated value of this factor at Key Lake is 2.4.

Handling Emissions. Details of the values used to calculate dust emissions for the handling emissions and the storage piles are shown in Table 5 of the main report. The production figures were calculated from the mining development plan which provides the number of trips for each size of vehicle. The volume of ore moved was adjusted to allow for mixing cobble ore into the mill feed.

The different variables were selected on the following basis:

- dumping capacity of various vehicles - from mining study;
- silt content - from EIS values;
- moisture content - estimated on basis of source, storage time, size of material;
- duration in storage - from production rate and total volume;

- silt content for open area - average of rock and sand values as they will be mixed, and
- erodibility - on the basis that it will approximate sand.

The total emission from handling is estimated to be 271 kg/a.

The dust emissions can be converted to radionuclide concentrations by assuming secular equilibrium, selecting an enhancement factor for small particles, utilizing the definition of activity (1.2×10^3 Bq U238/g U_3O_8), and knowing the degree of mineralization from each source. This conversion was utilized to produce the data used in the computer model.

A.2.1.2 Eldor Mines Atmospheric Emissions - Mining. As mining activities at the Rabbit Lake pit approach completion, the owner has initiated the development of a new mine at Collins Bay, approximately 9 km north of the existing site. The following calculations are based on a mid-1986 situation. The mine will be open pit with an ultimate size of 915 m x 305 m and a depth of 46 - 61 m. Mining will be on 1.5 - 3 m high benches about 61 m wide. For the purpose of radon estimates, an area of exposed ore of 2000 m^2 was selected for high grade ore (10-30% U_3O_8) mining.

In the projected eight-year mine life, 7.9×10^6 m^3 of waste (muskeg, overburden till, rock, and low grade ore) and 1.6×10^6 m^3 of ore will be mined. Most of the till overburden is to be removed in the first three years. Production will progress at 1.5×10^6 m^3/a for the first two years and 1.1×10^6 m^3/a for the balance of the mine life. The mill cutoff was selected as 0.03% U_3O_8; low grade ore will range from 0.03 - 0.074% U_3O_8. High grade ore (10-30% U_3O_8) will be blended with the lower grade material for milling grade averaging 0.45% U_3O_8. The primary waste will average 0.006% U_3O_8 with the majority at 0.001% or less.

The material mined from the B Zone deposit will be stored on site for subsequent haulage to the Rabbit Lake mine. Current plans call for three 50 000 t stockpiles, 6 m high and conical shaped, for average and low grade ore. Six additional high grade ore storage piles will be 1.5 m high and conical in shape.

The Rabbit Lake facility will be mined out by the end of 1984. The ore will be stockpiled for milling over the next two years. A total of 13×10^6 t of ore, or 7.3×10^5 m^3, will be stored. The pit will be used for tailings disposal from the B phase. Radon emission will occur from U_3O_8 mineralization in the exposed walls. The pit was approximated by a cone 127 m deep with a top diameter of 70 m and a base of 63 m.

Waste rock was assumed to have a mineralization of 0.006% U_3O_8, low grade 0.05% and "country" rock 0.001% U_3O_8.

Radon. The calculated areas and emission rates for both sites are outlined in Table 6 in the main report. The average radon flux rate outlined in the introduction was also utilized for this mine. The total radon generated at the Collins Bay site totals 34 TBq/a. This includes a contribution of about 0.7 TB/a from handling operations and mine dewatering. The mine dewatering is estimated to require a flow of 2050 L/min. The radon activity was taken at the maximum of 18.5 Bq/L. Ore handling includes 2 x 10^5 m^3/a of ore dumped and reclaimed at Collins Bay, 1.2 x 10^5 m^3/a of low grade waste dumped and 3 x 10^5 m^3/a of rock waste dumped. The till is assumed to be removed from the pit at this time.

At the Rabbit Lake pit, radon release rates predict emissions from only the waste dumps and the open pit walls. It is anticipated that part of the ore trucked from Collins Bay will be stored in piles on site near the mill. Handling releases were thus calculated to approximate one half the value for the Collins Bay site.

The Rabbit Lake pit will require dewatering during tailings disposal; this is not considered here. Water from dewatering the Collins Bay mine will be discharged at the Rabbit Lake site. It will offgas when discharged. Any water from the "country" rock around the pit will also offgas in the pit. It is estimated that the total release of radon from water indigenous to the Rabbit Lake site will be 0.04 TBq/a.

Dust Releases. Handling emissions occur at both Collins Bay and Rabbit Lake. It has been assumed that the ore handling emissions (Table 7 in the main text) are duplicated at Rabbit Lake. Factors for handling emissions are similar to those used for Key Lake in the absence of site-specific data. Since the B Zone deposit is partially under the lake, a moisture content of 10% was assumed.

Open source emissions occur at both sites. The ore pile at Collins Bay was assumed to be active: turnover of high grade - 5 days; of average grade - 156 days. Silt content was developed from KLMC figures. Open source emissions at Collins Bay are from the waste and special waste piles. The factors were again developed from the Key Lake figures. At Rabbit Lake, the active pile was assumed to be Collins Ore with a four-month supply (175 000 t).

Haulage. The last fugitive source that must be considered is the haulage of ore from B Zone to Rabbit Lake. Both road vehicular emissions and losses from vehicles must be considered. Approximately 353 000 t/a are hauled in 35 t trucks three-quarters full. Some 19 600 trips per year are necessary at 9.7 km each way. Speeds are estimated to vary with full (62 t) trucks averaging 30 km/h and empty (31 t) trucks averaging 50 km/h. Six wheels, a 5% silt content, and 250 dry days result in emission factors of:

- 1.94 k/km full
- 3.24 kg/km empty

A total emission of 9.9 x 10^5 kg/a is generated.

Losses from the trucks can be approximated from losses in coal transportation. PEDCO (1978) reports losses of 1.0 kg/t for a total of 343 t/a of ore lost. Guarnaschelli (1977) reports 0.11 kg/t•km losses for coal in trains. This would yield 3700 t/a of losses. The total loss of Uranium-238 could thus range from 37 x 10^6 Bq/a - 4.8 x 10^9 Bq/a.

A.2.1.3 Cluff Lake Atmospheric Emissions - Mining. The development at Cluff Lake is somewhat different from the others. A high grade deposit, the "D" ore body, has been mined out. The surface area of this pit is estimated at 11 000 m^2 of rock and 1300 m^2 of overburden. For estimating purposes, it is assumed that the surface will grade at 2 ppm U_3O_8.

Mining Operations. Subsequent to the completion of "D" zone mining, a test pit was created in the Claude ore body. This body along with the "N" ore body will be developed as open pits. In addition, three underground mines recover ore from the N40, OP, and Dominic Peter (DP) deposits. The current schedule calls for the Claude pit to operate to mid-1989 and the N pit from then to 1992. The OP underground mine will operate in 1985, N40 in 1986-87, and DP from mid-1987 to 1997. In exploiting these resources, the following volumes will be removed:

- Claude - 8 yrs. - 4.25 x 10^6 BCM and 5.8 x 10^5 t of ore @ 0.36%
- N - 5 yrs. - 2.13 x 10^6 BCM & 3.7 x 10^5 t @ 0.28%
- OP - 2 yrs. - 60 000 t ore @ 0.25% (125 tpd)
- N40 - 2 yrs. - 137 000 t ore @ 0.51% (240 tpd)
- DP - 10 yrs. - 1.2 x 10^6 ore @ 0.5% (480 tpd)

It is necessary to select a point in time to determine source levels so that the Cluff Lake facility can be treated in the same way as the other facilities. For the sake of comparison, mid-1987 was selected as representative. At that time, the short-lived underground mines will be completed, the Claude pit half excavated, and the development of the DP deposit well underway. This will result in the following facilities:

- Claude pit: 704 m x 340 m top, 670 x 270 m base, 45 m deep, 38° side walls
- overburden storage: 1.4 x 10^6 m^3
- waste storage: 2.2 x 10^6 m^3
- ore storage: 2.95 x 10^4 t or 1.64 x 10^4 m^3 (60 m x 60 m and approximately 6 m high)
- underground waste pile: 2.3 x 10^4 m^3
- ore storage pad for underground ore: similar to above
- ventilation rate from underground works (2.4 m x 2.4 m): 47 m^3/s.

Radon Emissions. Certain assumptions were made to calculate the radon emissions from the mine. The ore in the Claude pit is not in nugget form but rather in 3 layers. It was assumed that these were 10 m thick and 1/3 of the total ore was in each slice. This provides 2.3 x 10^4 m^2 of ore exposed @ 0.36% U_3O_8. Sub-ore was assumed to be of the same volume and uniformly distributed at any given depth. It was also assumed to contain mineralization at half the cutoff grade (0.015% U_3O_8). The balance of the exposed pit was assumed to be overburden to a mineralization of 2 x 10^{-4}% U_3O_8.

Waste and Ore Piles. Two Claude Pit waste dumps (ultimate size 2.3 and 3.5 x 10^6 LCM) will be created. Since only part of these will be utilized to the midlife of the Claude pit, it was assumed that they could cover the full base dimension but one would not attain the ultimate height. Furthermore, it was assumed that overburden cleared for the creation of the second waste pile would be used to cover the first pile and allow revegetation. Thus, one pile has sub-ore @ 0.015% U_3O_8, the other overburden @ 2 x 10^{-4}% U_3O_8. The heights of these piles are approximately:

- 2.3 x 10^6 LCM of rock waste, 20 m high and surface area of 1.14 x 10^5 m^2
- 1.4 x 10^6 LCM of overburden, 13 m high and surface area of 1.2 x 10^5 m^2

The ore storage pads are all the same. They are designed to store 29 500 t or 16 400 m^3 of ore. It was assumed that they are not full all the time but rather contain 12 300 m^3 of ore and have a total height of 6 m and a total area of 3200 m^2. The Claude pit will require 2450 m^3/d of dewatering and assuming 18.5 TBq/L of radium-226, will release 0.01 TBq/a of radon gas. The underground waste pile will contain 2.3 x 10^4 m^3 of waste in a pile of 53 x 73 x 6 m with a surface area of 5400 m^2.

Ventilation. The underground ventilation is designed to remove radon as it is generated. The IES (AMOK 1982) presents a radon generation rate of 1.7 x 10^3 Bq/s. The ventilation rate is 47 m^3/s for a radon discharge concentration of 36 Bq/m^3 and a total release of 0.05 TBq/a. Data from the Beaverlodge mine (Frost 1980) provide a value for dust releases from the underground mine. Concentrations underground were 370 x 10^{-6} Bq/m^3 of 238-U and other radionuclides. The dust release was assumed to be at the same concentration or 17.4 x 10^{-3} Bq/s.

Handling. Releases from the handling of materials were calculated on the basis of dumping the waste once and the ore three times (i.e. dumping ore on storage piles, reclaiming it and sending it to the crusher at the mill). Only the waste from the open pit was included and BCM (bank cubic metres) were converted to loose cubic metres (LCM) by a bulking factor of 1.6. The ore contains contributions for both the open pit at 59 100 m^3/a and underground at 5.5 x 10^5 m^3/a. Assuming ore is handled three times, the total handling release is 2.5 TBq/a.

A summary of radon emissions from AMOK is presented in Table 8 of the main text.

Dust. As with the other sites, handling and open source wind erosion contributes to fugitive dust generation. No haulage losses were calculated due to a lack of specific data and the relatively small quantities calculated at other sites. Handling losses were calculated for ore, waste, and reclaiming ore. The only active source considered are the ore stockpiles (Claude and underground). Only waste and overburden piles at Claude are considered. The unknown nature and small size of the underground waste piles precludes consideration. The dust emissions are summarized in Table 13 of the main report.

AMOK (owners of Cluff Lake) plan to use smaller trucks than those used at the other mines. Handling emissions are scaled accordingly. The waste silt content is projected at 15-18% for the overburden (AMOK 1976). A value of 10% was selected for this study since not all exposed waste will be overburden. Moistures were selected at the high end although the reclaimed ore is anticipated to dry somewhat.

For lack of better data, the active sources assume that ore will be of a similar nature to that at Key Lake. The storage time was estimated as one month since two piles will feed the mill. It has been assumed that ore storage at the mill will continue to be in closed facilities. No calculations have therefore been undertaken for this source.

For open sources, the factors used at the other sites are used at AMOK as well. The silt content was adjusted to reflect published data.

The dust emissions from underground mine ventilation are detailed in the mine radon section for AMOK.

A.2.2 Atmospheric Emissions - Milling Operations. This section will deal only with emissions from milling operations. Component sources are:

- crushing
- grinding
- leaching
- yellowcake packaging

Limited literature data is available to substantiate any emission factor chosen for these sources.

Dust Losses - All Mills. Several references elaborate on crushing losses. Sears et al. (1975) state that total dust losses can range from 1 x 10^{-4} percent to 5 x 10^{-4} percent of ore processed whereas NUREG-0706 uses a value of 0.4 g/t of material 1 μm or less. The Key Lake EIS (KLMC 1979) reports emissions of 0.95 kg/hr from the crusher, based on 720 tpd. This implies a factor of 0.03 kg/t of ore. Sears notes that if the base moisture content is taken as 6%, an increase in moisture of 33% (i.e., 8% moisture) drops the load to the control device by 87.5%. With typical control levels of 90-94%, ore of 9-10% may release only 20% of the amount that a controlled 6% moisture ore releases. Clearly some latitude is available in emission factors if moisture has this great an effect.

Neither the Rabbit Lake/Collins Bay B Zone EIS (Gulf 1980) nor the Cluff Lake Phase II EIS (AMOK 1982) provide data on crusher losses. No data were provided for these sources by either proponent when later requested by the study team. For lack of any other data, the value for Key Lake was selected for this study.

No data is available for grinding losses. This process is generally wet and therefore losses are minimal. In the absence of any other data, KLMC data was used to develop an emission factor.

Another source ignored in the literature is the leaching area. Here elevated temperatures and long cycle times can combine to release some dust. The KLMC factor of 2.1 x 10^{-3} kg/t ore was used to develop values for the other mills.

Yellowcake drying and packaging areas do get some consideration in all the EIS documents. This final product can ill afford to be released due to its value. NUREG-0706 cites an emission factor of 0.02% of production. This amounts to 0.2 kg/t of yellowcake. KLMC report an estimated 0.09 kg/t of product on the basis of better control devices. Rabbit Lake Mill reports (Gulf 1980) a new scrubber with a 99.8% efficiency will be installed. Based on the 0.02% of production loading to the scrubber, an estimated

4×10^{-5}% of production will be lost or 4×10^{-4} kg/t of product. AMOK (1982) make mention of high efficiency dust collectors but provide no emission data. On the basis of this data, the conservative KLMC factor was used for all mills. Dust losses for the mills are summarized in Table 10 of the main report. These are based on the following emission factors:

- crusher - 0.03 kg/t ore
- grinder - 0.02 kg/t ore
- leaching - 2.1×10^{-3} kg/t ore
- yellowcake - 0.09 kg/t product

The radionuclide emission rates for the various constituents were calculated on the basis of particulate emissions, and the accepted conversion factor of 1.2×10^{3} Bq/g for those radionuclides assumed to be in secular equilibrium. As U-238:Pb-210 ratio of 1:0.67 is unique to the Key Lake mine, (and reflects in the data presented in their EIS), this ratio was used for Kay Lake mine source terms. For the other mines, a 1:1 ratio was used. All particulates are assumed to have an activity enhancement of 2.4 over the bulk ore (NUREG-0706 1980).

Yellowcake emissions were assumed to have the following ratio of specific activities for the following constitutents: U-238:Th-230:Ra-226:Pb-210, 1:0.05:0.002:0.002 (NUREG-0706). This data is required for insertion into the dispersion and dosimetry code which requires each radionuclide to be listed separately.

Radon Emissions - All Mills. Radon emissions from the mill are presented in the AMOK (1976) and KLMC (1979) documents. These agree fairly well, given the grade of ore processed in the facilities. These values will fluctuate as a function of the amount of material in any given process stage, the emanation factor for the ore and the exact nature of the process. NUREG 0706 provides a method to calculate radon emisions from the mill. The equation presented is:

$$^{222}\mathrm{Rn} = E\,R_a \lambda T W$$

where $^{222}\mathrm{Rn}$ = radon released (Bq/a)
E = emanating power
λ = decay constant for Rn-222 (d^{-1})
T = cycle time (days)
Ra = concentration of Ra-226 in ore (Bq/g)
W = amount of ore processed (g/a)

The radium concentration is developed on the basis of secular equilibrium in the ore.

Generally the cycle time is in the order of 1.5 to 2 days. Emanation factors are 0.2 for Eldor and Cluff Lake (since no other data are available) and 0.3 for KLMC. On this basis, one can calculate radon released from the mill using this equation. The results are:

- Eldor - $0.2 \times 47 \times 0.181 \times 2 \times 2.2 \times 10^9 = 777 \times 10^6$ Bq/d
- Cluff Lake - $0.2 \times 37 \times 0.181 \times 2 \times 7.2 \times 10^8 = 185 \times 10^6$ Bq/d
- KLMC - $0.3 \times 260 \times 0.181 \times 2 \times 5.5 \times 10^8 = 1.6 \times 10^9$ Bq/d

Based on 7000 hours /year operating time this amounts to:

- Eldor - 222×10^9 Bq/a
- Cluff Lake - 56×10^9 Bq/a
- KLMC - 455×10^9 Bq/a

These numbers are at odds with the predicted KLMC release rate of 8.5 TBq/a (KLMC 1979). A check of radon emissions shows that at Key Lake, mine/mill sources contribute 86 TBq/a of radon to the atmosphere. The variation between the mill release numbers generated in this study and those in the Key Lake EIS is 8.1 TBq/a or 9% of the total release. This change would not produce significant dose changes. Since the radon release number for AMOK generated above agrees closely with that in their EIS, it was deemed appropriate to develop the value on a uniform basis, and this study's values will be used for modelling purposes. The distribution of the radon emissions will be as follows:

- crusher - 60%
- grinder - 5%
- leach - 35%

The radon source terms are summarized in Table 10 of the main report.

A.2.3 Emissions - Tailings Management Areas. The Uranium Dispersion and Dosimetry (UDAD) code provides a method of determining the tailings pile source strength for particulate emisions. MILDOS simplifies this procedure by limiting many of the variables to default options. For the sake of providing the reader with an understanding of the method, the following discussion of wind erosion from tailings is presented. Calculated radon source stengths are input into the program.

While wind erosion source terms in the previous section were developed on the basis of experimentally derived equations for storage piles, the UDAD code uses classical fluid mechanics principles combined with the work of Bagnold, Belly, Lettan, Gillette, and Travis (NUREG-0706 1980) to predict wind erosion source terms.

Particulate Losses From Tailings. Generally speaking, wind velocity profiles near the surface are described by a logarithmic equation as follows:

$$U_Z = U^* \, 2.5 \ln (Z/Zo) \quad (1)$$

where U_Z = wind velocity @ height Z (m/s)
U^* = shear velocity (m/s)
Z = height above the surface (m)
Z_o = surface roughness term determined to be the height where $U_Z = 0$ (m)

At some velocity the surface wind starts large particles moving and their exchange of momentum with particles of less than 500 μm size causes them to bounce along the surface, an effect called saltation. Particles that are larger than 100 μm return to the ground quickly due to gravity but smaller particles with settling velocities lower than the wind's turbulent velocity continue to travel for long distances. Bagnold and Belly developed an expression for the saltation threshold velocity as follows:

$$U_t^* = \sqrt{A \frac{\rho_s - \rho}{\rho} \quad gd \, (1.8 + 0.6 \log W)} \quad (2)$$

where: U_t^* = threshold value of the shear velocity (m/s)
s = particle density (g/m^3)
ρ = density of air (g/m^3)
g = gravitational constant (m/s^2)
d = average diameter of the grain (m)
A = dimensionless coefficient
W = water content % by weight.

The rate of horizontal particle movement is then described by:

$$\gamma = CU^{*2} (U^* - U_t^*) \quad (3)$$

where: γ = horizontal flux in (g/cm-s)
C = 10^{-6} (g-s^2/cm^4)

The vertical flux for wind pickup of fine (less than 20 µm) particles is defined by:

$$V_{20} = C_V (U^*/U_t^*)^{\alpha} \tag{4}$$

where V_{20} = vertical flux of less than 20 µm material (mass/unit time and unit area)

α = constant that is a function of the fraction of fine particles

C_V = proportionality constant (2×10^{-6} g/m^2·s)

Since the vertical flux is associated with a horizontal flux, Travis developed a formula based on the relationship of V_{20} and γ:

$$V_{20} = \gamma (C_V/U_t^{*3} C_h) ((U^*/U^*_t)^{P/3} - 1) \tag{5}$$

where C_h = 10^{-2} g.sec^2/m^4

P = particle mass percentage less than 20 µm in diameter.

It is Travis' form of V_{20} that is used in MILDOS. The minus one (-1) in the above equation deals with the physical limit of V_{20} that occurs when no material less than 20 µ m in size is present in the source.

For the particulate material having a diameter d 100µm in MILDOS, the total vertical flux V for radionuclide i, in activity is estimated from

$$V_i = V_{20} I_{20i}/F_{20} \tag{6}$$

where I_{20i} = specific activity of radionuclide i with d 20 µm (Bq/g)

F_{20} = the activity fraction of suspended particulates that are less than 20 µm in diameter.

The specific activity for particles with diameter less than 20 µm is obtained from the bulk specific activity of all particles times an activity ratio factor (2.5).

For each wind speed class, the release of radioactive particulates from a contaminated area is calculated for the average wind speed. The annual release is the summation of each wind speed class contribution.

Many of the variables discussed above are defaulted in MILDOS. The following is a list of the default variables:

- surface roughness - Zo 1 cm
- particle density - s = 2.4 g/cm^3
- average grain diameter - d = 0.03 cm

- water content - $w = 0.1\%$
- mass less than $20 \mu m$ - $P = 3\%$
- activity fraction of less than 20 μm material - $F_{20} = 0.5$
- small particle specific activity enhancement factor = 2.5.

Since no facility is available to adjust these variables on MILDOS, they were utilized. Actual site values estimated from data supplied by the operators is provided in Table 11 of the main report. A consideration of the variation in source strength is presented in the discussion.

In the following pages, the mill tailings sites are discussed in terms of the additional values used for the variables in the above equations. Several added variables are needed to account for the northern Saskatchewan weather conditions. The most important factor is some allowance for the climate of the region. For a simple approach, it was decided to assume that wind erosion occurred all year. In actual fact, the tailings areas will likely be frozen for between 180 and 200 days per year.

The dry beach area is the only area subject to wind erosion. Since this area is likely to move within the confines of the tailings area, it was decided to scale the total area of the pond to respresent the dry beach area. The wind erosion scale factor modifier, shown in Table 11 of the main report, accounts for the assumed dry beach area.

Radon Emissions From Tailings. Radon emission calculations can be made on the basis of the Fickian diffusion equation, simplified to assume uniform radium distribution and a thickness greater than 3 to 4 m (NUREG-0706). In this simple case, the specific radon flux in Bq/m^2-s is given by:

$$J = Ra\, E\rho\, (\lambda\, D/P)^{1/2} \times 10^4$$

where Ra = radium activity (Bq/g)

E = emanating power, 0.2

ρ = tailing density, 1.6 g/cm^3

λ = $2.1 \times 10^{-6}/s$

D = effective bulk diffusion coefficient (cm^2/s)

P = porosity (%).

The D/P ratio is influenced by moisture in the tailings and this unknown factor makes it necessary to assume some value.

The generic impact statement (NUREG 0706 1980) references the work of Haywood et al. and quotes a range of specific radon flux from infinitely thick tailings piles. These values are 0.013 Bq Rn/m^2·s per Bq Ra/g for wet tailings, 0.024 for moist, and 0.044 for dry tailings. On a conservative basis, the summer flux was chosen to correspond to the moist figure of 0.024 Bq Rn/m^2·s per Bq Ra/g. This provides a D/P value of 1.96 x 10^{-2}. Countess (1977) presents radon flux measurements for tailings which show average values of 144 Bq/m^2·s; these range from3.7Bq/m^2·s for dry tailings to 1.5 Bq/m^2·s with ice cover. Since the tailings will be mostly frozen in winter, an arbitrary emission level equal to 10 percent of the summer emission was postulated.

The D/P ratio, otherwise defined as k, is shown by Tanner (1965) to vary between 5.4 x 10^{-2} cm^2/s for sand with 4 percent moisture to 2.2 x 10^{-6} for mud with 85 percent moisture. Sears (1975) states that the diffusion coefficient in air is 10^4 times that in water. On the basis of this data, it is possible to neglect the contribution of radon from tailings that are underwater. Radon from the tailings was thus calculated on the basis of the dry beach area and it was assumed that the tailings were frozen for 180 days/a.

It should be noted that the MILDOS code was run for the yearly case and no allowance was made for differing summer and winter source strengths, although the total yearly radon release was input into the model.

A.2.3.1 Key Lake Tailings Area Emissions. The tailings at Key Lake will be impounded in an approximate 50 ha area. The Key Lake tailings will generally be deposited using the sub-aerial technique. In this technique, the tailings are discharged through spray bars or open trough launders so that the slurry flows gently over a beach to form a slurry layer approximately 100 μm thick. The beaches form at a natural gradient of approximately 1 in 100. When a section of the beach has been covered with a slurry, the discharge is stopped and the slurry is left exposed for a period before the next layer is placed on top of it. During this time, the solids in the slurry settle and the excess effluent bleeds off and drains away to the supernatant pond at the lower end of the beach. During settling, some segregation takes place in the solids and the fine clay and silt-sized particles are concentrated at the surface to form a relatively impervious, cohesive crust. This gives rise to very low releases for both particulates and radon from the Key Lake tailings area.

KLMC provided many of the factors included in the UDAD wind erosion calculations (Table 11 of the main report) although defaults were used for MILDOS. A wind erosion scale factor of one-third was chosen as most representative of this site.

The radon emission for Key Lake was based on the EIS value of 11 x 10^6 Bq/s over the 50 ha area used in that report. Since this value was based on experimental data it is postulated that it will only apply during the summer period. As with the other sites, 10 percent of this value was used to calculate the emission total for the winter period. The total quantity is then input into the programme.

A.2.3.2 Eldor Mines Tailings Area Emissions. There will be two tailings sites at Rabbit Lake. The existing facility is a conventional tailings pond with an ultimate size of 50 ha. Forty percent of this will be assumed to be dry beach area. Gulf (1980) report data from this tailings area showing 25-80% moisture in borehole samples, an average of 33% of the borehole material at less than 20 microns and 56% less than 100 μm. The saltation diameter is likely less than 100 μm given the preponderance of this material. MILDOS default values for suface roughness, activity fraction of the fines, and particle density will be used.

The moisture and size of the material will greatly affect the radon emission rate. The variation in these factors affect the D/P value and thus the flux. Data in Sears (1975) show that an addition of 3% moisture halves the effective D/P ratio. This would imply D/P = 1.05×10^{-4} for 30% moisture and D/P = 2.1×10^{-6} for 80% moisture. The latter results in a flux of 2.5×10^{-4} Bq/m^2·s/Bq [Ra-226]. KLMC (1979) postulate 1.7×10^{-4} Bq/m^2·s/Bq [Ra-226]. Since the deposition method is different, a value of 1×10^{-4} cm^2/s for D/P was selected for this site, corresponding to 30% moisture. The flux is thus 31×10^{-6} Bq/m^2·s based on 47 Bq/g radium. With 20 ha, this provides 100×10^6 Bq released in the summer and 100×10^5 Bq released in the winter. This totals 110×10^6 Bq/a released.

The second tailings area will be in the abandoned Rabbit Lake pit. This has been designed to be flooded at all times, thus minimizing radon releases and eliminating fugitive dust releases. No allowance is provided for this tailings area in the present study.

A.2.3.3 Cluff Lake Tailings Area Emissions. The existing tailings facility at Cluff Lake is currently designated for enlargement to handle Phase II tailings (AMOK 1982). It is anticipated that a large portion of this area will be flooded. In the absence of more detailed information, MILDOS default values were selected, only 20% is assumed to be exposed and the particle mass fraction was selected at 20% to compare favourably with the other sites.

All tailings data is summarized in Table 11 of the main report.

A.2.4 Emission Source Terms in MILDOS Modelling. The MILDOS code provides many standard default options. Although Appendix A1 discusses parameters related to environmental transport and dose calculations, some air dispersion values should be discussed at this time. The handbook for MILDOS (NUREG/CR-2011) and the final Generic EIS (NUREG-0706) discuss the modelling approach of the code and for a detailed discussion the reader is referred to these publications. Of interest at this time are the effects of climate and terrain on the code results.

Plume Buoyancy. The user's guide states that the MILDOS code employs a Holland plume rise formulation without the thermal buoyancy term:

$$h = (1.5\ Vd)u$$

where h = plume rise above stack exit (m)
V = exit velocity (m/s)
d = stack diameter (m)
u = wind speed (m/s)

This is not strictly an appropriate equation, both in terms of the code or the physical realities of northern Saskatchewan.

The MILDOS computer programme provides no means to input a stack diameter, so the plume rise value is low if the stacks are greater than 1 m in diameter and the plume rise is high if the diameter is less than 1 m. The justification, likely used by the authors but not stated, for ignoring thermal effects is the relatively small difference between stack gas temperatures and the ambient temperature in the southern U.S. This might be true in New Mexico but with average temperatures of -4°C in northern Saskatchewan, 30° or 40°C temperature spreads can lead to significant thermal plume rises. Briggs (1974) denotes plume rise near the stack as:

$$h_r = C_1 F^{1/3}\ X^{2/3} u$$

where C_1 = a dimensionless constant
F = buoyancy flux
X = downwind distance (m)
u = wind speed at stack height (m/s)

and $F = r^2 V_s\ G((Tg-Ta)/Tg)$

where r = stack radius (m)
V_s = stack gas exit velocity (m/s)
G = gravitational constant (m/s^2)
Tg = stackgas exit temperature (°K)
Ta = ambient temperature (°K)

Obviously, the buoyancy flux can vary from 0 (for Tg = Ta) to some finite value, but the minimum differential temperature acceptable for most purposes is 1°C. Assuming an average mill temperature of 20°C, this adds 20% to the buoyancy flux term. Modifying the code to account for this term involves Briggs fits for stable, neutral, and unstable conditions and is beyond the scope of this work.

Since many of the open wind erosion sources at these sites will probably not emit dust during the winter, yearly joint wind speed, wind direction, and stability data are not strictly applicable but were employed. Similarily the source strength was assumed to be uniform throughout the year. As was mentioned in previous sections, an attempt was made to scale seasonal radon emissions to provide a yearly uniform release rate. No attempt was made to run monthly stability data to examine the effect of differing source strengths and wind conditions.

Area Source Size. For open sources (area sources), the model converts the area into equivalent square of width "d". It is assumed that a virtual point source is located at a distance of d/2 cotan θ/2 upwind from the centre of the source area, where θ is the 22.5° sector used to subtend the area width. This is a standard approach for modelling area sources but is costly to run if "d" becomes too small, since many calculations must be undertaken for each source element. For this reason, larger values of "d" were selected to encompass the area to be modelled and the calculated concentrations for nearby receptor locations may be somewhat uncertain.

Tailings areas were generally split into 4 sub-areas for modelling.

Receptors. Receptor locations were specified for both identified populated areas and on other points of interest. Due to low populations in northern Saskatchewan, radial increments were used, thereby reducing computing time. Population data for each run were based on 1976 census statistics. Population data are summarized in Table C.5. It was assumed that all fishing/hunting camps have an average yearly population of 10 persons.

STAR Data. Standard joint frequency data from the Atmospheric Environment Service were used where possible. This STAR data were from Uranium City for Cluff Lake and from Cree Lake for the Key Lake mine. For the sites near Wollaston Lake, a composite Collins Bay set was developed using Cree Lake stabilities and Collins Bay wind records. Generally, an average mixing depth of 400 m was assumed for all sites.

APPENDIX B

AQUATIC MODELLING DEVELOPMENT

B.1 ENVIRONMENTAL TRANSPORT MODELS AND PARAMETERS

The aquatic dispersion model used in this study was developed for uranium mill tailings disposal sites and has been described in other reports (Beak 1983a). It incorporates radionuclide release from the tailings management area, dilution by uncontaminated water and sedimentation processes in contaminated rivers, lakes, and reservoirs. A linear concentration factor approach is used to model radionuclide transfer from water through the food chain. Sensitivity analysis (Beak 1983a) has shown that sedimentation parameters such as particle concentration, settling velocity, and water column adsorption coefficients are very important in prediction of radionuclide dispersion and consequent dose impact. These parameters are discussed in Setion B.1.2 below.

B.1.1 Aquatic Emission Sources Terms. Tailings leaching calculations were not performed in the present assessment, since source values were obtained from monitoring data obtained from small streams downgradient of the tailings management areas under study. These monitoring loadings would thus contain short time frame (< 20 year) seepage loadings. Discharge from the tailings treatment area is considered in all cases to be controlled, and assumed constant over the 15-year operational period. Radionuclide concentrations in the discharge, as determined by monitoring over a period of years, are given in Table 3. The total uranium concentration is considered to consist of U-238 and U-234 in secular equilibrium. Emissions from the tailings management area are assumed for modelling purposes to cease completely at the end of the operational period.

B.1.2 Aquatic Dispersion of Radionuclides. Transport of radionuclides through the aquatic environment depends on their dilution and physical dispersion in downstream lakes, streams, and reservoirs, and on their biotic uptake and transfer from one living organism to another. The physical parameters of downstream water bodies which control radionuclide dispersion are listed in Table B.1 Flow patterns are summarized in Figures 3, 4, and 5. Volumes, discharge rates, and retention times are based on hydrological data compiled in 1977 by the Inland Waters Directorate (1978a). Surface area was determined by planimetry from topographic maps and used to calculate mean lake depth where this data was not directly available. Sediment area was assumed equal to surface area for calculation of sediment volume. Activity calculations assume instantaneous mixing of radionuclides as they enter each lake. The continuous progression of radionuclides through the lake system was approximated by repeated calculations at discrete 1-year time intervals.

TABLE B.1 PHYSICAL PARAMETERS OF DOWNSTREAM LAKES

Lake Name	Surface Area (m^2)	Mean Depth (m)	Volume (m^3)	Discharge Rate (m^3/a)	Retention Time (a)	Sediment Volume (m^3)	
						Aerobic	Anoxic
Key Lake System							
Russell	4.39E7	13.5	5.94E8	7.25E8	1.08	1.76E6	4.39E6
Keefe	1.94E7	20.5	3.98E8	1.10E9	0.37	7.76E5	1.94E6
No name	5.08E6	23.0	1.17E8	1.32E9	0.08	2.03E5	5.08E5
Nekweaga Bay	8.70E7	8.8	7.61E8	2.34E9	0.33	3.48E6	8.70E6
Wollaston*	2.08E9	20.4	4.24E10	5.55E9*	14.00	8.32E7	2.08E8
Reindeer	6.64E9	14.3	9.47E10	1.10E10	11.00	2.66E8	6.64E8
Hatchet	1.50E8	13.9	2.09E9	8.20E8	2.50	6.00E6	1.50E7
Black	4.66E8	14.9	6.96E9	9.46E9	0.72	1.86E7	4.66E7
Athabasca	7.94E9	25.8	2.05E11	4.73E10	6.00	3.18E8	7.94E8
Cluff Lake System							
Old Fort Bay	6.68E1	1.0	6.81E7	8.20E8	0.08	2.67E6	6.68E6
Athabasca	7.94E3	25.8	2.05E11	4.73E10	6.00	3.18E8	7.94E8
Rabbit Lake System							
Otter bay	8.62E7	9.0	7.72E8	2.57E8	3.00	3.45E6	8.62E6
Wollaston*	2.08E9	20.4	4.24E10	5.55E9*	14.00	8.32E7	2.08E8
Reindeer	6.64E9	14.3	9.47E10	1.10E10	11.00	2.66E8	6.64E8
Hatchet	1.50E8	13.9	2.09E9	3.60E9	2.50	6.00E6	1.50E7
Black	4.66E8	14.9	6.96E9	9.46E9	0.72	1.86E7	4.66E7
Athabasca	7.94E9	25.8	2.05E11	4.73E10	6.00	3.18E8	7.94E8

*Discharge from Wollaston is to Reindeer (96.6%) and Hatchet (3.4%) Lakes.

Radionuclides are partitioned within each lake between the water, the suspended particulate fraction and the sediments. A suspended particle concentration of 0.4 mg/L, comprised of equal parts clay, silt, and organic matter, is assumed. Sedimentation of adsorbed radionuclides is calculated from their water column adsorption coefficients (Table B.2) and a particle settling velocity of 25 m/year. This velocity is typical of the organic particulate fraction in mass balance models (Snodgrass and O'Melia, 1975; Dillon and Kirchner, 1975; Thomann et al., 1977; Ng, 1981) and corresponds to a sediment accumulation rate of 0.02 cm/year. Kipphut (1978) reported sediment accumulation of 0.07-0.8 cm/year in several Canadian Shield lakes which he studied.

Binding within the sediments is based on sediment adsorption coefficients (k_d). These coefficients vary widely with physical and chemical sediment properties, such as pH, temperature, and redox potential. Thus the typical k_d values used in the model (Table B.2) should not be considered representative of any particular site. Ideally, site-specific measurements should be obtained. Water-column adsorption coefficients (below) were not found in the literature, and were assumed equal to those which characterize the sediment. This procedure is conservative since it probably underestimates water column adsorption and radionuclide loss by sedimentation. O'Connor and Connelly (1980) suggest that adsorption coefficients are inversely related to particulate concentration, and should actually be greater in the water column than in the sediment.

TABLE B.2 RADIONUCLIDE ADSORPTION COEFFICIENTS (mL/g)

	Model Values		Observed Values	
Radionuclide	Water*	Sediment	Range	References
U-238, U-234	10^4	10^4	16-270	Rancon, 1973
Th-228, Th-230, Th-232	$5x10^3$	$5x10^3$	1-10^6 40-130	Randon, 1973 Nishiwaki et al., 1972
Ra-226, Ra-228	500	500	150-3000 214-467	Kirshmann et al., 1966 Serne et al., 1977
Pb-210, Po-210	10^3	10^3	-	

* Water column adsorption coefficients are assumed equal to sediment values; their true values may be greater (O'Connor and Connelly 1980).

Most sediments accumulate in the deep depositional zone of lakes and are continually buried by new sediment. Thus, surficial sediments will have a specific activity proportional to the activity of particulates in the overlying water column. Deeper sediments will be anoxic and have an activity corresponding to water activity at the time of deposition, corrected for decay and diffusional losses to overlying sediment. Surficial sediments, 4 cm in depth (Kipphut 1978), and deep sediments, 10 cm in depth, were assigned density and porosity values typical of Canadian Shield lakes (Table B.3).

TABLE B.3 SEDIMENT CHARACTERISTICS

Characteristic		Model Value	Observed Values
Accumulation Rate (cm/a)		0.02	0.07-0.80
Porosity (%)	Aerobic	95	94-97
	Anoxic	88	83-94
Density (g/cm^3)	Aerobic	1.02	1.02
	Anoxic	1.10	1.10
Depth (m)	Aerobic	4.0	4.5
	Anoxic	10.0	-

Radionuclides can diffuse from either sediment layer back into the overlying water column. Any radionuclides buried deeper than 14 cm are unlikely to influence diffusion across the sediment-water interface and are considered to be lost from the system.

Radionuclides may be incorporated into shoreline beach sediments to become a potential source of external radiation exposure. For modelling purposes, a 1% transfer to beaches was assumed. After radionuclide emission ceases, shoreline erosion and continuing sedimentation ensure that contaminated sediments are eventually buried. Calculated changes in water and sediment radionuclide concentrations over time for lakes downstream of tailings management areas are given in Tables D.1 and D.2, Appendix D.

B.1.3 Transfer of Radionuclides to Foods. Biological uptake of radionuclides from sediment or water results in their transfer through the food chain. Aquatic plants tend to accumulate radionuclides so that tissue concentrations exceed those of the surrounding water. Animals feeding on those plants may acquire even greater tissue concentrations depending on the particular element being considered. Radionuclide transfer from one link in the food chain to another is described by a concentation factor. These factors (P_{ij}) are defined by the ratio of radionuclide concentration in the consuming organism (C_j) to that in the food itself (C_i). Under steady-state conditions, both C_j and C_i are constant and the transfer coefficient is given by:

$P_{ij} = C_j/C_i$

Water-food concentration factors (transfer coefficients) have been determined for many human food types (Thompson et al. 1972). These are defined by the steady-state ratio of radionuclide concentration in human food to that in water. Since there are many potential routes (r) for radionuclide transfer from water to a particular food, and many biotic compartments (k) along each route, the concentration factor is given by:

$$P_{wf} = C_w(\sum_{rk} \pi \; P_{ij})$$

where π indicates serial multiplication of concentration factors between adjacent compartments i and j.

For aquatic modelling purposes, only the major foods harvested from northern Saskatchewan waters have been considered. Concentration factors for wild rice and fish are shown in Table B.4. Moose and caribou are the primary big game animals of the region. Aquatic plants are a minor component of moose diet, and caribou diet is entirely terrestrial. Both mammals are considered in the atmospheric model.

TABLE B.4 CONCENTRATION FACTORS (Bq/kg per Bq/L)

Radionuclide	Water	Wild Rice*	Fish
U-238	1	20	10
U-234	1	20	10
Th-230	1	30	30
Ra-226	1	50	50
Pb-210	1	4	100
Po-210	1	40	50

* Rice concentration factor is 2% of that for whole aquatic plants (Thompson et al. 1972).

APPENDIX C

DETAILED RADIOLOGICAL IMPACT CALCULATIONS
ATMOSPHERIC PATHWAYS

C.1 DOSES TO HUMANS FROM ATMOSPHERIC PATHWAYS

From airborne radioactivity concentrations and ground contamination levels generated by MILDOS, radiation doses to human individuals and to populations will be calculated in this section using models and parameters developed in Appendix A.1 and dosimetry data given in Section 4 of the main report.

Detailed estimates of individual doses will be made for only one site (Rabbit Lake) in order to provide an understanding of the general magnitude of the impact on individuals and to illustrate the assessment methodology. It is not the intention of this study to carry out three detailed assessments for all the residents in the regions surrounding the three identified mine sites. Doses are evaluated after 15 years of operation. Rabbit Lake was selected on the basis of having the highest population concentration.

Estimates of collective population doses from atmospheric emissions were made for populations within 80 km (50 miles) of all three identified mine sites, to assess the regional impacts from uranium mining activities in northern Saskatchewan. This group is called the 'Airshed Population'. See definitions in Section 4 of the main report.

C.1.1 Air Concentrations and Inhalation Doses. The effective dose equivalent commitments from the inhalation of radon and particulates are given in Table C.1 for identified populated locations. Doses from individual particulate radionuclides at Wollaston Lake are given in Table C.2. Radionuclide concentrations in air at populated locations are given in Table C.3. Dose conversion factors are obtained from Tables 14 and 15 of the main report.

The calculated inhalation doses to the most exposed individuals at the Wollaston Lake Community (considered to be the "Critical Group") are 25.3 μSv/a from radon, and 2.1 μSv/a from particulates.

C.1.2 Ground Contamination and External Doses. External doses result from exposure to contaminated ground and air. Air concentrations are discussed in the previous section. Radionuclide concentrations on the ground are given in Table C.4. External doses from immersion in air and from ground depositions are given in Tables C.1 and C.5. External dose conversion factors used are as given in Tables 14 and 15 of the main report. The MILDOS calculations assume a building shielding factor of 0.5 and that the individual spends 60 percent of the time indoors. As a conservative assumption in the present study, the building shielding factor was set equal to 1.

TABLE C.1 MAXIMUM ANNUAL INDIVIDUAL DOSE FROM RADON AND PARTICULATES AT SELECTED AIRSHED LOCATIONS WITHIN 80 km OF A MINE SITE

	Inhalation (Sv/a)		External (Sv/a)	
Location[1]	Radon[2]	Particulate	Radon[2]	Particulate[3]
Hidden Bay	0.456E-4	0.127E-4	0.341E-5	0.304E-5
Estevan Island	0.202E-4	0.319E-5	0.148E-5	0.690E-6
Wollaston Lake	0.253E-4	0.204E-5	0.183E-5	0.430E-6
Waterbury Lake	0.965E-5	0.700E-6	0.710E-6	0.146E-6
Waterfound Lake	0.966E-5	0.570E-6	0.720E-6	0.120E-6
Unknown Lake	0.769E-5	0.430E-6	0.570E-6	0.890E-7
Geike River	0.665E-5	0.720E-6	0.491E-6	0.920E-7
Russell Lake	0.528E-4	0.609E-5	0.386E-5	0.145E-5
Pendleton Lake	0.200E-4	0.170E-5	0.154E-5	0.257E-6
Cree Lake	0.134E-4	0.690E-6	0.101E-5	0.207E-6
Mawdsley Lake	0.106E-4	0.710E-6	0.780E-6	0.170E-6
Fleming Lake	0.112E-4	0.520E-6	0.850E-6	0.102E-6
Middleton Lake	0.100E-4	0.520E-6	0.890E-6	0.101E-6
Brander Lake	0.924E-5	0.119E-5	0.678E-6	0.179E-6
Old Fort Bay	0.220E-5	0.270E-6	0.166E-6	0.381E-7

[1] Full year occupancy on location is assumed.
[2] Includes radon daughters.
[3] Mainly from ground exposure.

The calculated total individual dose from external radiation to the critical group (at Wollaston Lake) is 2.3 μSv/a.

C.1.3 Doses from Ingestion of Vegetation. It is assumed that at Wollaston Lake, the nearest major populated area around the Rabbit Lake site, edible vegetables are grown in backyard gardens. Approximately half of northern Saskatchewan has a frost-free period of more than 60 days, which should be enough to support such garden crops. Radioactivity levels in vegetation are calculated using the transfer model and parameter values in Appendix A, Section A.1.4. Doses from ingestion of vegetables are calculated assuming an individual consumption rate of 50 kg/a fresh weight and using ingestion dose conversion

TABLE C.2 SUMMARY OF MAXIMUM ANNUAL INDIVIDUAL DOSE (mSv/a) FROM AIRBORNE PARTICULATES AND DEPOSITED RADIONUCLIDES AT WOLLASTON LAKE

	U-238	U-234	Th-230	Ra-226	Pb-210	Po-210	Total
Internal							
Ingestion	0.159E-4	0.179E-4	0.124E-4	0.807E-3	0.174E-3	0.574E-2	0.677E-2
Inhalation	0.430E-3	0.480E-3	0.103E-2	0.300E-4	0.500E-4	0.300E-4	0.204E-2
External							
Air	0.179E-9	0.810E-11	0.449E-11	0.211E-10	0.252E-10	0.268E-13	0.238E-9
Ground*	0.400E-4	0.500E-5	0.300E-5	0.370E-3	0.400E-6	0.900E-6	0.459E-3
Total	0.486E-3	0.503E-3	0.105E-2	0.121E-2	0.264E-3	0.577E-2	0.928E-2

* Pb-210 and Po-210 values include ingrowth from radon.

factors given in Table 14 of the main report. Air and soil concentrations (after 15 years of deposition) are obtained from MILDOS outputs (Tables C.3 and C.4). Results of these calculations are shown in Tables C.2 and C.6. Vegetation contamination levels result almost entirely from aerial deposition on foliage with very little contribution from soil uptake. The calculated individual dose from ingestion of vegetables at Wollaston Lake is 0.4×10^{-3} μSv/a.

C.1.4 **Doses from Ingestion of Meat.** The complex nature of the "food web" operating in a terrestrial ecosystem has been discussed in Section 3 of the main report. In northern Saskatchewan, the more important terrestrial food webs as far as radiological impact on humans is concerned are:

1) (lichens, grasses, herbs)-to-caribou-to-man
2) (grasses, herbs, shrubs, woody barks, aquatic plants)-to-(moose, deer, beaver, muskrat, other land mammals)-to-man.

Caribou Diet. Studies (Miller 1976b) have shown that lichens are the primary forage of barren-ground caribou on the taiga winter range in northern Saskatchewan. Analysis of caribou rumen contents show that about 80 percent of the caribou's diet consists of lichens during November, declining to about 50 percent in April when northward spring migration to calving grounds begins. During the summer months, the

TABLE C.3 CONCENTRATION OF RADIONUCLIDES IN AIR AT SELECTED AIRSHED LOCATIONS WITHIN 80 km OF A MINE SITE

	Concentration (Bq/m^3)							
Location	U-238	U-234	Th-230	Ra-26	Pb-210	Po-210	Rn-222	Rn Daughters (WL)
Hidden Bay	2.2E-5	2.2E-5	1.1E-5	9.9E-6	9.9E-6	9.9E-6	1.0E+0	2.44E-3
Estevan I.	5.0E-6	5.0E-6	2.6E-6	2.4E-6	2.4E-6	2.4E-6	4.4E-1	0.113E-3
Wollaston L.	3.1E-6	3.1E-6	1.7E-6	1.7E-6	1.7E-6	1.7E-6	5.4E-1	0.143E-3
Waterbury L.	1.0E-6	1.0E-6	5.9E-7	5.7E-6	5.7E-6	5.7E-6	2.1E-1	0.548E-4
Waterfound L.	8.3E-7	8.3E-7	4.9E-7	4.7E-6	4.7E-6	4.7E-6	2.1E-1	0.549E-4
Unknown L.	6.1E-7	6.1E-7	3.7E-7	3.5E-7	3.5E-7	3.5E-7	1.6E-1	0.437E-4
Geike R.	6.7E-7	6.7E-7	3.8E-7	3.6E-7	3.6E-7	3.6E-7	1.4E-1	0.378E-4
Russell L.	8.9E-6	8.9E-6	5.4E-7	5.1E-6	3.4E-6	3.4E-6	1.1E+0	0.296E-3
Pendleton L.	1.4E-6	1.4E-6	1.0E-6	9.7E-7	6.5E-7	6.5E-7	4.4E-1	0.118E-3
Cree L.	1.0E-6	1.0E-6	6.1E-7	5.8E-7	3.9E-7	3.9E-7	2.9E+0	0.764E-4
Mawdsley L.	9.8E-7	9.8E-7	6.5E-7	6.3E-7	4.3E-7	4.3E-7	2.3E-1	0.603E-4
Fleming L.	7.5E-7	7.5E-7	4.6E-7	4.4E-7	2.9E-7	2.9E-7	2.4E-1	0.638E-4
Middleton L.	7.9E-7	7.9E-6	4.5E-7	4.4E-7	2.8E-7	2.8E-7	2.5E-1	0.673E-4
Brander L.	2.1E-6	2.1E-6	8.7E-7	7.7E-7	7.7E-7	7.7E-7	2.0E-1	0.521E-4
Old Fort Bay	4.6E-7	4.6E-7	2.0E-7	1.8E-7	1.8E-7	1.8E-7	4.7E-2	0.125E-4

caribou's diet is more varied, consisting of grasslike plants, twigs, leaves, and mushrooms in addition to lichens.

Caribou Consumption. Studies of seasonal movements and distribution of caribou in the Canadian north (Parker 1972; Thomas 1969) show that barren-ground caribou spend 5 to 6 months a year (November to April) at their winter grazing ground below the tree line. During this period, regional caribou herds may approach the mining sites. These are hunted by the region's inhabitants. In recent years, caribou hunting has diminished due to diminishing availability of herds and also the fact that the natives' existence no longer depends entirely on these animals, as it once did. Estimates made more than 10 years ago (Thomas 1969) showed that during the autumn and winter of 1966-67, about 3000 caribou were killed in the entire Stony Rapids, Fond du Lac, and Wollaston

TABLE C.4 CONCENTRATION OF RADIONUCLIDES IN SOIL AT SELECTED AIRSHED LOCATIONS WITHIN 80 km OF A MINE SITE

	Concentration* (Bq/m^2)					
Location	U-238	U-234	Th-230	Ra-226	Pb-210**	Po-210
Hidden Bay	64	64	42	42	50	50
Estevan I.	14	14	9	9	19	19
Wollaston L.	9	9	6	6	3	3
Waterbury L.	3	3	2	2	1	1
Waterfound L.	2	2	2	2	2	2
Unknown L.	2	2	1	1	1	1
Geike R.	2	2	1	1	1	1
Russell L.	25	25	20	20	44	44
Pendleton L.	4	4	4	4	34	34
Cree L.	3	3	2	2	26	26
Mawdsley L.	3	3	2	2	16	16
Fleming L.	2	2	1	1	25	25
Middleton L.	2	2	1	1	22	22
Brader L.	6	6	2	2	9	9
Old Fort Bay	1	1	1	1	3	3

* After 15 years of mining operations.
** Pb-210 and Po-210 concentrations include ingrowth from Rn-222.

Lake areas. Kill figures at the present time are probably lower still, and it is considered unlikely even for the more heavy consumers of caribou meat at these locations to consume much more than 50 kg/a fresh weight. All of the ingestion amounts referred to in this section unless otherwise indicated are expressed as fresh weights.

For the purpose of impact assessment, the radionuclide contents of caribou grazing entirely on lichens were calculated at all populated locations within 80km of a mine site. Radinuclide concentrations in lichens and transfer from lichen to caribou meat are calculated using the model and parameter values in Appendix A, sections A.1.5 and A.1.6.

Air concentrations are obtained from MILDOS output. A deposition period of 15 years is assumed. As discussed in Section A.1.5, an observed air-to-lichen transfer

TABLE C.5 MAXIMUM ANNUAL INDIVIDUAL DOSE AND COLLECTIVE DOSE COMMITMENT AT SELECTED AIRSHED LOCATIONS WITHIN 80 km OF A MINE SITE

	Dose (Sv/a)			Collective Dose (person-Sv/a)	
Location	Internal	External	Population*	Internal	External
Hidden Bay	0.992E-4	0.645E-5	25	0.248E-2	0.161E-3
Estevan I.	0.364E-4	0.217E-5	(10)	0.364E-3	0.217E-4
Wollaston	0.341E-4	0.226E-5	460 (82)	0.185E-1	0.123E-2
Waterbury L.	0.127E-4	0.856E-6	(10)	0.127E-3	0.856E-5
Waterfound L.	0.122E-4	0.835E-6	(10)	0.122E-3	0.835E-5
Unknown L.	0.954E-5	0.658E-6	(10)	0.954E-4	0.658E-5
Geike R.	0.887E-5	0.583E-6	(10)	0.887E-4	0.583E-5
Russell L.	0.740E-4	0.531E-5	0	0	0
Pendleton L.	0.246E-4	0.179E-5	(10)	0.246E-3	0.179E-4
Cree L.	0.158E-4	0.122E-5	22	0.348E-3	0.268E-5
Mawdsley L.	0.132E-4	0.954E-6	(10)	0.132E-3	0.954E-5
Fleming L.	0.130E-4	0.954E-6	(10)	0.130E-3	0.954E-5
Middleton L.	0.118E-4	0.995E-6	(10)	0.188E-3	0.995E-5
Brander L.	0.136E-4	0.857E-6	0	0	0
Old Fort Bay	0.320E-5	0.204E-6	0	0	0

* Transient populations (in parentheses) are based on camp capacities and occupancy factors from the Midwest Lake Environmental Baseline Study (Beak 1980) and on population data from the Saskatchewan Hospital Services Plan, 1982.

factor of 2.1 x 10^{+5} m^3/kg (fresh weight) is used for Pb-210. For the other radionuclides, radionuclides, a derived transfer factor of 7 x 10^5 m^3/kg is used. Assuming a lichen ingestion rate of 7 kg/d (fresh weight), transfer to caribou meat is calculated using transfer data in Table 23 of the main report. Dose to human individuals from ingesting caribou meat is calculated assuming a meat consumption rate of 50 kg/a and using ingestion dose conversion factors given in Tables 14 and 15 of the main report. Results of the above calculations for Wollaston Lake are summarized in Table C.6. Dose to an individual consuming caribou meat using the above assumptions is 6.8 μSv/a.

TABLE C.6 RADIONUCLIDE CONCENTRATIONS IN FEED AND MEAT AND DOSE TO HUMANS BY INGESTION AT WOLLASTON LAKE

	U-238	U-234	Th-230	Ra-226	Pb-210	Po--210	Total
Concentration (Bq/kg)							
Vegetation	1.6E-3	1.6E-3	9/7E-4	1.2E-3	1.2E-3	8.7E-4	7.5E-3
Forage	1.6E-3	1.6E-3	9.7E-4	2.8E-2	4.6E-3	8.7E-4	1.2E-2
Grass	3.4E-3	3.4E-3	2.0E-3	2.2E-3	4.7E-3	1.9E-3	1.8E-2
Lichen	2.1E+0	2.1E+0	1.2E+0	1.2E+0	3.3E-1	1.2E0	8.1E0
Aquatic Plant	1.0E-1	1.0E-1	1.4E-3	6.4E-3	2.4E-4	2.3E-3	2.2E-1
Moose	9.6E-5	9.5E-5	9/6E-6	4.0E-5	8.3E-5	5.5E-4	8.8E-4
Caribou	5.1E-3	5.1E-3	1.7E-3	5.4E-2	2.6E-3	2.7E-2	3.4E-1
Beaver	4.1E-5	4.1E-5	4.4E-7	4.1E-6	2.4E-6	3.3E-5	1.2E-4
Human Dose (Sv/a)							
Vegetables	0.137E-10	0.155E-10	0.193E-10	0.440E-10	0.285E-9	0.640E-10	0.440E-9
Moose	0.608E-11	0.688E-11	0.140E-11	0.121E-10	0.110E-9	0.259E-9	0.395E-9
Caribou	0.158E-7	0.179E-7	0.124E-7	0.807E-6	0.172E-6	0.574E-5	0.677E-5
Beaver	0.232E-10	0.261E-10	0.590E-12	0.130E-10	0.350E-10	0.146E-9	0.244E-9
Ingestion	0.159E-7	0.179E-7	0.124E-7	0.807E-6	0.173E-6	0.574E-5	0.677E-5

Moose, deer, and other gamehunting is carried out by residents and tourists. These game are fairly evenly and sparsely distributed in northern Saskatchewan. For the present study, the transfer of radionuclides to moose as representative big game and subsequent intake by humans is evaluated. Doses from consumption of beaver are estimated for the critical group.

Moose Diet. It consists of grasses, forbs, shrubs, tree foliage and aquatic plants in summer, and the twigs and woody barks of willow, balsam fir, red-osier redwood, ash, aspen, and birch in winter. The food requirements are approximately 17 kg wet weight (4.8 kg dry weight) per day (Velovsky et al. 1975). In summer, the daily intake of aquatic plants is about 6 kg wet weight (0.6 kg dry eight). Averaged over the year, aquatic plant intake would be about 2 kg/d wet weight (0.2 kg/d dry weight). The water intake rate is assumed to be the same as that of cattle, mammals of comparable size, i.e., about 50 L/d.

The radionuclide uptake of moose browsing in the vicinity of Wollaston lake will be calculated. It is assumed that these moose obtain their drinking water and aquatic plants from Wollaston Lake. Lake water concentrations are calculated as discussed in Appendix B. They are given in Table 20 of the main report and aquatic plant concentrations are calculated using bioaccumulation factors given in Table 19 of the main report. Radionuclide concentrations in forage grasses, herbs, and shrubs in the vicinity of Wollaston lake are calculated as described in Appendix A. Air and soil concentrations are obtained from MILDOS printouts. Calculated forage concentrations are given in Table C.6.

Transfer of radioactivity from diet to moose meat is determined by the animal feed-to-meat transfer factors F (Bq/kg per Bq/d intake in feed) as given in Table 23 of the main report. The moose diet consists of 3 components: land forage (27 kg/d wet weight), aquatic plants (2 kg/d wet weight), and water (50 L/day). The concentration in moose meat C (Bq/kg) is given by:

$$C_m = (27\ C_v + 2\ C_p + 50\ C_w)\ F_m$$

where C_v = concentration in forage (Bq/kg wet wt)

C_p = concentration in aquatic plants (Bq/kg wet wt)

C_w = concentration in water (Bq/L)

Calculated moose meat concentration and doses to humans from consuming moose meat are summarized in Table C.6. It is assumed that for the critical group (native

Indians), the average annual consumption of moose meat is about 1 kg. The estimated ingestion dose is 4.0×10^{-3} µSv/a. Although aquatic plants constitute less than 10% of the moose diet, plants from contaminated water are a major contributor to radioactivity levels in moose meat. This is due to large bioaccumulation factors of aquatic plants for radioactivity used in this study.

Other Game. Apart from large mammals such as moose, caribou, and deer, small mammals such as beaver, muskrat, lynx and snowshoe rabbit are also hunted and consumed, particularly by native Indians. It has been estimated (Shindelka 1978) that the consumption of small fur animals in the Churchill River Basin amounts to 19 to 48 kg/a, averaging about 30 kg/a. Beaver and muskrat are the major species consumed. The radionuclide content of beaver in the Wollaston Lake area and ingestion doses to humans was estimated.

Beaver is an amphibious animal with a semi-aquatic habitat and is considered to be representative of the small mammals in the area. The average weight of beaver is about 20 kg. It consumes about 0.7 kg/d of woody vegetation such as leaves, twigs, buds, tree barks (Banfield 1974). This is assumed to be supplemented by 1 kg/d of aquatic plants. The water intake rate is assumed to be 2 L/day. Using the same procedure as that used for moose, and feed-to-meat transfer factors given in Table 23, it may be shown that the ingestion dose to humans from consuming 10 kg/a of beaver in the Wollaston Lake area is about 2.0×10^{-3} µSv/a (Table C.6). As is the case with the moose, the bulk of radioactivity in beaver is derived from aquatic plants.

C.3.1.5 Collective Doses to Airshed Population. Collective doses from inhalation and from external exposure for populations within 80 km (50 miles) of the three mine sites are calculated. Since northern Saskatchewan is very sparsely populated, the following simple procedure is used:

1) identify all populated locations within 80 km (the "Airshed Populations");
2) estimate size of population in each location;
3) calculate airborne concentrations, external air immersion, and ground dose rates at each location (obtained from MILDOS run outputs);
4) calculate doses to individuals at each loation; and,
5) multiply individual doses by population size to obtain collective dose.

Results of these calculations are summarized in Tables C.5 and C.7. As would be expected for such a sparsely populated region, the collective doses are small,

TABLE C.7 MAXIMUM ANNUAL COLLECTIVE DOSE COMMITMENT TO AIRSHED POPULATIONS (person-Sv/a)

	Cluff Lake	Key Lake	Rabbit Lake*	Total
Internal				
Ingestion	0	0.108E-3	0.521E-2	0.532E-2
Inhalation	0	0.864E-3	0.181E-1	0.187E-1
External				
Air	0	0.600E-4	0.115E-2	0.121E-2
Ground	0	0.143E-4	0.359E-3	0.373E-3
TOTAL	0	0.105E-2	2.48E-2	2.59E-2

* Includes estimated doses to transient populations at Minor Bay (10), Salaba Lake (5), Moral Lake (5) and Michael Lake (5).

amounting to 0.026 person-Sv/a of which 72% is from radon by inhalation or external exposure. Due to population distribution, the bulk of the collective doses is from Rabbit Lake/Collins Bay releases.

Populations sizes of permanent populations are taken from recent census data. Transient population sizes are estimated from the capacity of existing fishing camps and a peak occupancy factor of 78%. This represents a conservative assumption since camp occupancy will usually be much lower.

C.2 RADIATION DOSES TO TERRESTRIAL BIOTA

C.2.1 Introduction. As is the case with humans, radiation exposure pathways for terrestrial animals are:

1) external irradiation
2) inhalation
3) ingestion

Limitations in Biota Dose Assessment. Apart from external irradiation doses, which are calculated in a manner essentially identical to that for humans, dose calculations for terrestrial animals are highly uncertain due to the lack of appropriate metabolic and dosimetric data for animals or reliable experimental radionuclide transfer data from environmental media to animals. Terrestrial biota doses calculated in this study are in general based on conservative assumptions and should be regarded as upper limit estimates.

Significance of Biota Doses. An extensive body of data exists on the effects of ionizing radiation on biota. However, very few general conclusions have been made, particularly regarding acceptable dose levels. In contrast to the situation for humans, the fate of an individual organism of a particular species is considered only of importance insofar as it has consequences at the population level, i.e., through the loss of the species' ability to survive and reproduce in competition with all the other biological components of the particular ecosystem (IAEA 1979). The dose level at which a population or community effect becomes manifest is difficult to quantify.

The Bayda Commission (1978) recorded evidence that if man is protected, no apparent effects on vegetation or animals will be observed. Studies carried out at the Whiteshell Nuclear Research Establishment in Pinawa, Manitoba, demonstrated experimentally that a dose rate of 0.035 Gy/a has no observable effects on plant species; at a dose rate of 0.24 Gy/a, no adverse radiation effects were observed; at a dose rate of 0.31 Gy/a, stimulation in growth was observed; a dose rate of 26 Gy/a was lethal (Bayda Commission 1978). Calculated doses to plant species may be compared against these observations.

C.2.2 External Irradiation of Terrestrial Biota. External absorbed doses from immersion in contaminated air and from contaminated ground are assumed to be the same as that calculated for humans, except that in the case of species other than man, a building shielding factor is not used. Following procedures similar to those used in Section 4 of the main report, but disregarding the building shielding factor, external whole body dose rates calculated for locations in the vicinity of the Rabbit Lake site are shown in Table C.8. External absorbed doses from contaminated ground are calculated at a height of 1 m above ground. For small animals foraging close to the ground, the dose rates may be somewhat higher. The difference in dose rate is small for gamma radiation and may be ignored. For beta radiation, which delivers doses primarily to skin, the difference in skin dose rate between 1 m and 0.2 m above contaminated ground could be one to two orders of magnitude in the case of humans, depending on the beta energy. However, the depth of the sensitive layer of skin in animals may be assumed to be larger than that for humans. It is also well recognized that the radiosensitivity of skin is very low. For these reasons, skin irradiation is not calculated for animals. Radioactive materials may also deposit directly on the fur and skin of animals and primarily irradiate skin. Again, for the reasons stated above, these doses are not calculated.

TABLE C.8 EXTERNAL DOSE RATES TO BIOTA IN THE VICINITY OF HIDDEN BAY (μGy/hr)

	Air Immersion	Ground Contamination	Total
U-238	0.117E-9	0.727E-5	0.727E-5
U-234	0.166E-11	0.940E-6	0.940E-6
Th-230	0.117E-11	0.791E-6	0.791E-6
Ra-226	0.150E-10	0.272E-3	0.272E-3
Rn-222*	0.357E-3	-	0.357E-3
Pb-210	0.436E-11	0.350E-5	0.350E-5
TOTAL	0.358E-3	0.282E-3	0.640E-3

* Includes daughters Po-218, Pb-214, Bi-214.

C.2.3 **Inhalation by Land Mammals.** Due to substantial physiological differences between humans and animals, the lung dynamics model and dose equivalent weighting factors developed for humans cannot be used here. Sufficient information is not available to warrant a generalized consideration of this pathway for all animal species. Inhaled radioactive materials lead to irradiation of internal organs, as well as the lungs. For terrestrial animals, internal organ doses from inhalation of particulates are expected to be small compared to doses from ingesting contaminated food and therefore will not be calculated. However, lung doses from inhalation of radon daughters may be significant.

Radon dosimetry is complicated. In the case of humans, the basal cells of the bronchial epithelium are considered to be significantly affected. The dosimetry is strongly influenced by factors such as the fraction of unattached daughters, aerosol particle size, diameter and surface areas in different regions of the lung, and depth of the sensitive basal cells. In the case of animals, it is not clear whether the critical tissue is the same as that for humans. Detailed parameter values for proper dosimetry modelling are unavailable. No lung irradiation figures are calculated for mammals.

The exposure rate to radon daughters for mammals at Wollaston Lake can be crudely estimated at about 0.007 WLM/a. This is based on a constant inhalation rate equivalent to that of a working man. Actual inhalation rates will vary considerably among mammalian species. Experimental data on lung cancer induced in rats by inhaled radon, reviewed by the ICRP (ICRP 31 1981), showed a lung tumor incidence rate of 10 percent at an exposure level of 200 WLM. It is very unlikely at the radon exposure rates in the vicinity of mine sites to expect significant adverse effects in animals.

C.2.4 **Ingestion by Land Mammals.** Ingestion pathways for animals are:

- ingestion of contaminated food;
- ingestion of contaminated water;
- ingestion of radioactive particulates during the normal daily grooming of the animal's fur.

The last pathway is difficult to model, probably not significant, and will not be considered. The general steps used in evaluating doses to terrestrial animals are:

1. calculate radionuclide concentrations in the animal's diet and in water;
2. calculate radionuclide concentrations in various organs and tissues of the animal by:
    a) using experimental diet-to-animal tissue transfer factors, or
    b) using animal metabolic models and data;
3. calculate absorbed dose rates in organs and tissues.

The methodology used in calculating radionuclide concentrations in animal diet and water has been discussed in Appendix A and in Appendix B. Calculation of the transfer from diet to animals should ideally be carried out using animal metabolic data and suitable radionuclide metabolic models. For example, if metabolism follows first order kinetics, i.e., if the retention of radionuclides in a given organ or tissue is characterized by a single biological retention half life T, the radionuclide concentration in a particular organ or tissue is given by:

$$C_m = \frac{f_i I}{\lambda_b M}(1-e^{-\lambda bt}) \qquad (C.1)$$

At equilibrium:

$$C_m = \frac{f_i I}{\lambda_b M} \qquad (C.1a)$$

where C_m = radionuclide concentration in the organ or tissue (Bq/kg)

f_i = fraction of ingested radionuclides reaching the organ of interest

I = ingestion rate (Bq/d)

λ_b = first order biological elimination constant for the organ = $0.693/T_b$ (day^{-1})

M = mass of the organ or tissue (kg)

T_b = biological retention half life (days)

t = time (days)

Since $I = C_f Q_f$ where C_f is the concentration in the feed or diet (Bq/kg) and Q_f is the daily intake rate of feed (kg/d), the ratio of the concentration in the organ or tissue to that in the diet is:

$$C_m/C_f = f_i Q_f/\lambda_b M \qquad (C.2)$$

Since the daily intake rate Q_f is roughly proportional to the organ or body mass M, C_m/C_f will be approximately constant if f_i and b are constant. For a given radionuclide, f_i may be assumed to be approximately the same for most animal species. However, the biological retention half life tends to decrease (and λ_b tends to increase) with decreasing body mass, although not linearly (Reichle et al. 1970). Hence, if C_m/C_f ratio obtained for large mammals is applied to small mammals, the calculated concentration in the small mammals will likely be somewhat high, i.e., conservative. This is the case when C_m/C_f data obtained for caribou and other large mammals are used for beaver (see Appendix A.1).

Suitable metabolic data required for Equation C.1 for the various animal species are rarely available. Some attempts have been made to use parameter values accepted for humans (LeClare et al. 1975). This is usually unsatisfactory, particularly when dealing with bone seeking radionuclides whose metabolism usually cannot be described by a single retention half life. Direct experimental diet-to-tissue transfer data are preferred if they are available.

Once the radionuclide concentrations in the organs or tissues have been obtained, dosimetry calculations are relatively straightforward. For alpha and beta radiations, essentially all the decay energy is absorbed in the organ or tissue, and the dose rates are given by Loevinger et al. (1956).

Alpha Radiation

$$D = 5.8 \times 10^{-4} E\, C \ \mu\mathrm{Gy/h} \qquad \text{(C.3)}$$

where E = the alpha energy per distintegration (MeV)

C = the radionuclide concentration (Bq/kg)

Beta Radiation

$$D = 5.8 \times 10^{-4} \bar{E}\, C \ \mu\mathrm{Gy/h} \qquad \text{(C.4)}$$

where $\bar{E}$ = the average beta energy per disintegration (MeV)

Because gamma rays are far more penetrating than alpha or beta particles, the absorbed gamma dose is related not only to the gamma energies but also to the size and shape of the organ or tissue containing the radioisotope. The absorbed gamma dose rate is given by Loevinger et al. (1956).

Gamma Radiation

$$D = 2.7 \times 10^{7}\, \Gamma C P \bar{g} \ \mu\mathrm{Gy/h} \qquad \text{(C.5)}$$

where Γ = the Specific Gamma Ray Constant which relates activity and exposure ($\mathrm{cm}^2 \cdot \mathrm{R/mCi \cdot h}$ at 1 cm)

C = the radionuclide concentration (Bq/kg)

P = the tissue density (taken to be 1 $\mathrm{gm/cm}^3$)

$\bar{g}$ = the average geometrical factor (cm)

The average geometrical factor g is a factor that takes into account the particular size and shape of the organ, as well as gamma energy absorption in the organ. It may be shown (Loevinger et al. 1956) that for radionuclides uniformly distributed in a sphere of radius r cm,

$$\bar{g} = 3r$$

This is a fairly good approximation for radius $r < 10$ cm. For animals or organs whose shape may be approximated by a cylinder of a given radius and length, g factors may be obtained from Table IV of Leovinger et al. (1956 p. 858).

Values of E_α, $\bar{E}_\beta$ and Γ are calculated from radionuclide data in the Radiological Health Handbook (1970).

C.2.5 Dose Estimates for Selected Terrestrial Biota

C.2.5.1 Lichens. Lichens are selected as the critical terrestrial plant species because doses received by them are likely to be the highest among plant species in the area. They have large sorption areas, live for many years, and are very efficient collectors of atmospheric fallout. As discussed in Appendix A.1, direct transfer factors from air to lichen are 2×10^5 m^3/kg for Pb-210 and 7×10^5 m^3/kg for the other radionuclides. Radiation doses received by lichens growing at Hidden Bay are calculated using air concentration data obtained from MILDOS outputs and Equations C.3, C.4, and C.5 in Section C.2.5. For gamma dose calculations, lichens are assumed to be represented by a sphere of radius 5 cm. The average geometrical factor $g = 3\pi r = 47$ cm. Results of the dose calculations are shown in Table C.9.

The estimated dose to lichens at Hidden Bay is 2.1 mGy/a.

C.2.5.2 Mammals - Caribou, Moose, Beaver. Dose calculations have been carried out as described for caribou, moose, and beaver in the vicinity of Hidden Bay. Results are summarized in Table 21 of the main report. These show that the higher doses are received by caribou, a consequence of its assumed heavy diet of lichens. The organ receiving the highest dose is bone, again expected since most of the radionuclides involved are bone seekers. As a general conclusion, it may be stated that the maximum bone dose received by mammals is less than 3.6 mGy/a and that no single organ other than bone is likely to receive a dose in excess of 10 mGy/a.

Large mammals such as caribou roam over large areas and are probably less susceptible to local contamination close to the mine site than indicated by these calculations. The calculations are, therefore, conservative. The air-lichen-caribou pathway is likely to dominate the dose to man as long as caribou are assumed to remain within an 80 km radius of the mine site. Radionuclide concentrations in the meat of caribou and moose at Wollaston Lake have been estimated as shown in Table C.6. Concentrations in the flesh of beaver are similarly calculated assuming daily intakes of

TABLE C.9 DOSE RATES TO BIOTA FROM INCORPORATED RADIONUCLIDES IN THE VICINITY OF HIDDEN BAY (μGy/hr)

	Lichens	Beaver	Caribou	Moose
Uranium*				
Alpha	0.155E+0	0.452E-4	0.369E-3	0.471E-4
Beta	0.147E-1	0.428E-5	0.349E-4	0.446E-5
Gamma	0.179E-5	0.185E-7	0.256E-6	0.399E-7
Th-230				
Alpha	0.200E-1	0.107E-1	0.280E-4	0.210E-6
Ra-226				
Alpha	0.188E-1	0.107E-5	0.880E-3	0.250E-4
Gamma	0.164E-6	0.333E-9	0.464E-6	0.162E-8
Pb-210 (+ Daughters)				
Alpha	0.587E-2	0.605E-7	0.451E-4	0.876E-6
Beta	0.518E-3	0.534E-8	0.398E-5	0.838E-7
Gamma	0.320E-7	0.118E-10	0.148E-7	0.380E-9
Po-210				
Alpha	0.206E-1	0.651E-6	0.475E-2	0.198E-4

* Includes U-238, Th-234, Pa-234, U-234.

1 kg of aquatic plants from Wollaston Lake and using diet-to-meat transfer factors given in Table 23 of the main report.

From the concentrations in meat, radiation doses to muscle are calculated using equations C.2, C.4 and C.5. For gamma dose calculations, the animals are represented by cylinders with the following dimensions:

caribou	20 cm radius x 140 cm length
moose	30 cm radius x 200 cm length
beaver	10 cm radius x 50 cm length

The corresponding $\bar{g}$ values are 180 cm, 220 cm, and 106 cm.

Doses to other organs and tissues are estimated from data on the distribution of radionuclides in the body. The distribution of Pb-210 and Po-210 in caribou has been studied in some detail (Persson 1972, 1973). The average natural Pb-210 concentrations found in bone were 190 Bq/kg, in kidney 11 Bq/kg, in liver 26 Bq/kg, in blood 1.2 Bq/kg, and in meat 0.5 Bq/kg. Thus, the concentration ratios are bone: kidney: liver: blood: muscle = 400:23:54:2.5:1. Po-210, in contrast to all the other natural alpha emitters, is not a bone seeker but rather accumulates in soft tissue. Its presence in bone is primarily derived from the decay of Pb-210 in situ. From Persson's (1972) data the following concentration ratios for Po-210 are derived - bone: kidney: liver: blood: muscle 18:18:14: 1.1: 1.

Ra-226 is an alkaline earth radionuclide and, like Sr-90, is expected to follow the behaviour of calcium in the animal's body. Data obtained for Sr-90 in caribou (Persson 1971) show that the Sr-90/Ca ratios in bone and in muscle are approximately the same. It would seem reasonable to assume that the Ra-226 distribution in the various organs and tissues may be derived from the calcium distribution. The calcium contents in various soft animal tissues (Diem and Lentner 1970), in mg per kg, are: 110-200 (beef), 110 (beef kidney), 70 (beef liver), 100 (horse flesh), 50-120 (pork), 180 (rabbit meat), and 120-150 (poultry). The levels are fairly constant in soft tissue, with a mean value of about 150 mg/kg. The calcium content in bone is about 10% (UNSCEAR 1977), or 100,000 mg/kg. Therefore, if Ra-226 follows Ca, one would expect a bone-to-soft tissue concentration ratio of about 100 000/150 = 660.

The Sr-90 data for caribou (Garner 1972) indeed show a bone-to-muscle concentration ratio of 350. It may be deduced from data on the distribution of Ra-226 in swine (Nelson and Rust 1967) that, on a wet weight basis, the bone-to-muscle concentration ratio for Ra-226 is about 260 and that the concentrations of Ra-226 in different soft tissues such as kidney, liver, and muscle are within a factor of 2 of each other. Based on the above data, a bone-to-soft tissue concentration ratio of 300 will be assumed for Ra-226. Actual field data from northern Saskatchewan (Lush and Swanson 1981) suggest that the bone-to-soft-tissue ratio for fish varies with the species under investigation but ranges from about 100-150.

From data given for humans (UNSCEAR 1977), the concentration of U-238 in soft tissues ranges from 1 to 10 Bq/kg, while those in bone are from 150 to 200 Bq/kg. These give a bone-to-soft tissue concentration ratio of 15 to 150. Concentrations given for Th-232 are 50 Bq/kg for bone and 2 Bq/kg for soft tissue, a ratio of 25. For the present purpose, a ratio of 100 will be assumed for both U and Th. It is known that U accumulates in kidney, although not to the same extent as in bone. The concentration ratio of kidney or liver to muscle is assumed to be 30 for U and Th.

APPENDIX D

DETAILED RADIOLOGICAL IMPACT CALCULATIONS AQUATIC PATHWAYS

D.1 DOSES TO HUMANS FROM AQUATIC PATHWAYS

Doses received by individual members of the public from a given release will depend on their age and habits, as well as the radionuclide concentrations in their environment. In order to take these cultural factors into account, ICRP (1977) recommends that a critical population group, typical of the one expected to receive the highest dose, be defined in detail. This group should be culturally homogeneous and large enough to permit statistical evaluation of its homogeneity. Precise criteria for critical group homogeneity have not been given.

The Wollaston Lake population was defined as the critical group for this assessment. It is the first permanent settlement downstream of the Key Lake and Rabbit Lake mine sites. Critical group calculations were also performed for the Lake Athabasca population, including communities from Fort Chipewayan east to Uranium City. These are the first permanent settlements downstream of the Cluff Lake mine site, and were expected to receive lower radionuclide concentrations than the Wollaston Lake community. Discharge from Old Fort Bay was assumed to mix completely with the rest of Lake Athabasca before flowing west into the MacKenzie River system. Critical group calculations include infant and adult doses. Doses to regional populations downstream of the critical group were calculated for adults only.

D.1.1 Radionuclide Concentrations in Water and Sediment. Radionuclide concentrations in surface waters are shown in Table D.1 at intervals during and after the operational period. These are calculated incremental concentrations due to tailings discharge and do not include natural background levels. Water concentrations of released radionuclides increase rapidly after discharge begins, and reach their equilibrium values within a few years. These values are maintained throughout the remainder of the operational period. Water concentrations decline with time in a negative exponential fashion after discharge ceases at 15 years.

In general, the highest water concentrations are found in lakes closest to and downstream of a tailings discharge. This is not a function of actual distance, but of the dilution which occurs with distance as uncontaminated water enters the system. It is also a function of sedimentation which removes radionuclides from the system as they move downstream. The greatest radionuclide concentrations for the sites modelled occur in Otter Bay and Russell Lake, and the lowest in Reindeer Lake. Peak concentrations in

TABLE D.1 RADIONUCLIDE CONCENTRATIONS IN WATER (Bq/L)

	Year*					
	1	5	15	17	20	30
Russell Lake						
U-238	1.1E-2	1.1E-2	1.1E-2	4.1E-5	6.3E-8	5.0E-8
U-234	1.1E-2	1.1E-2	1.1E-2	4.1E-5	6.4E-8	5.0E-8
Th-230	1.4E-4	1.4E-4	1.4E-4	5.1E-7	1.6E-9	1.4E-9
Ra-226	3.0E-4	3.0E-4	3.1E-4	1.2E-7	2.9E-8	2.7E-8
Pb-210	1.3E-4	1.3E-4	1.3E-4	5.1E-7	9.5E-9	1.1E-8
Po-210	1.5E-4	1.5E-4	1.5E-4	2.6E-7	9.5E-9	1.1E-8
Wollaston Lake						
U-238	9.3E-5	1.0E-4	1.0E-4	9.3E-5	1.9E-7	1.6E-9
U-234	9.7E-5	1.1E-4	1.1E-4	9.5E-6	2.0E-7	1.6E-9
Th-230	7.0E-7	9.2E-7	9.2E-7	7.2E-8	1.4E-9	2.6E-11
Ra-226	2.5E-6	2.6E-6	2.6E-6	2.5E-7	6.0E-9	8.1E-10
Pb-210	9.5E-7	1.2E-6	1.2E-6	3.5E-7	2.6E-9	3.0E-10
Po-210	9.0E-7	1.1E-6	1.1E-6	6.8E-8	1.8E-11	3.0E-10
Reindeer Lake						
U-238	2.7E-6	3.3E-6	3.3E-6	4.6E-7	1.1E-8	7.0E-11
U-234	2.8E-6	3.4E-6	3.4E-6	4.7E-7	1.2E-8	7.1E-11
Th-230	2.0E-8	2.4E-8	2.4E-8	3.5E-9	9.2E-10	1.1E-12
Ra-226	7.2E-8	8.6E-8	8.6E-8	1.2E-8	3.7E-10	3.5E-11
Pb-210	2.8E-8	3.4E-8	3.4E-8	5.0E-9	1.4E-10	1.4E-11
Po-210	2.8E-8	3.3E-8	3.3E-8	3.8E-9	1.0E-10	1.4E-11
Lake Athabasca						
U-238	7.5E-6	8.4E-6	8.4E-6	8.7E-7	2.9E-8	9.1E-11
U-234	7.8E-6	8.6E-6	8.6E-6	9.0E-7	3.0E-8	9.4E-11
Th-230	1.7E-8	1.9E-8	1.9E-8	1.9E-9	6.7E-10	4.3E-13
Ra-226	1.0E-7	1.1E-7	1.1E-7	1.2E-8	4.1E-10	2.4E-11
Pb-210	3.5E-7	3.9E-8	3.9E-8	4.3E-9	1.6E-10	8.6E-12
Po-210	6.3E-7	6.6E-8	6.6E-8	3.1E-9	1.1E-10	8.5E-12

* After commencement of operation with close-out in year 15.

Reindeer Lake are four orders of magnitude lower than those in Otter Bay and Russell Lake.

Wollaston Lake receives radionuclides from both Key Lake and Rabbit Lake sites. Based upon monitored values from Eldor Resources and projected values from Key

Lake, the greatest uranium and radium contributions come from the Eldor Resources site, while the Key Lake site contributes the greater portion of thorium, lead, and polonium. Uranium concentrations predominate, exceeding those of the other nuclides by two orders of magnitude. The incremental uranium concentration here represents 1.7% of natural background in Wollaston lake water (Beak 1983b).

Lake Athabasca receives radionuclides from all three sites, but only those from the Cluff Lake site make a significant contribution. Contributions from the other two sites are four orders of magnitude lower.

Radionuclide concentrations in sediments are shown in Table D.2 at intervals up to 30 years from the beginning of the 15-year operational period. Whereas peak concentrations in water were observed during the operational period, peak sediment concentrations occurred during the post-operational period. This is because net release of radionuclides from sediments by diffusion into the water column does not occur until water concentrations have fallen several orders of magnitude from their operational levels. For uranium, thorium, and radium in Wollaston Lake, this happens at 18 years, 3 years after tailings discharge has ceased. Peak sediment concentrations of lead and polonium were not reached during the 30-year time frame of this study, though the accumulation rate was less than 1% per year at 30 years. Ingrowing of lead and polonium from radium decay contributes to the continued accumulation of these radionuclides in sediments.

The greatest sediment concentrations of radionuclides occur in those lakes with the highest water concentrations. Peak sediment concentrations in Reindeer Lake are four orders of magnitude lower than those in Otter Bay and Russell Lake, as expected from the water concentrations of radionuclides in these areas.

In Wollaston Lake, the greatest sediment loadings of uranium and radium come from the Rabbit Lake site, while the Key Lake site is the major contributor of thorium, lead, and polonium. As in the water column, uranium loadings predominate by two orders of magnitude. The incremental uranium concentration in sediments of Wollaston Lake represents approximately 2% of natural background in lake sediments from northern Saskatchewan (Hornbrook and Garrett 1976).

Radionuclide concentrations in Lake Athabasca sediments are higher than those in Reindeer Lake, and are almost entirely due to input from the Cluff Lake site.

These results suggest that at the end of the operational period, careful monitoring of surficial sediments in Russell Lake and Otter Bay may reveal detectable changes in surficial sediment concentrations. Beyond these receiving waters,

TABLE D.2 RADIONUCLIDE CONCENTRATIONS IN SEDIMENT (Bq/kg)

	Year					
	5	10	15	20	25	30
Russell Lake						
U-238	3.2E+1	6.4E+1	9.6E+1	1.0E+2	9.8E+1	9.5E+1
U-234	3.3E+1	6.6E+1	9.8E+1	1.0E+2	1.0E+2	9.8E+1
Th-230	4.E-1	8.7E-1	1.3E0	1.4E0	1.3E0	1.3E0
Ra-226	9.2E-1	1.8E-1	2.7E0	2.9E0	2.8E0	2.7E0
Pb-210	4.6E-1	9.8E-1	1.6E0	1.8E0	1.9E0	2.0E0
Po-210	4.9E-1	1.0E0	1.6E0	1.8E0	2.0E0	2.1E0
Wollaston Lake						
U-238	2.9E-1	6.1E-1	9.1E-1	9.8E-1	9.8E-1	9.4E-1
U-234	3.0E-1	6.3E-1	9.4E-1	1.0E0	9.8E-1	9.6E-1
Th-230	2.2E-3	4.6E-3	6.8E-3	7.4E-3	7.3E-3	7.1E-3
Ra-226	7.7E-3	1.6E-3	2.1E-2	2.6E-2	2.5E-2	2.4E-2
Pb-210	3.3E-3	7.8E-3	1.3E-2	1.5E-2	1.7E-2	1.8E-2
Po-210	3.7E-3	8.1E-3	1.3E-2	1.6E-2	1.7E-2	1.8E-2
Reindeer Lake						
U-238	8.6E-3	1.8E-2	2.8E-2	3.0E-2	2.9E-2	2.9E-2
U-234	8.9E-3	1.9E-2	2.8E-2	3.1E-2	3.0E-2	3.0E-2
Th-230	6.3E-5	1.4E-4	2.1E-4	2.3E-4	2.3E-4	2.2E-4
Ra-226	2.3E-4	4.8E-4	7.3E-4	7.9E-4	7.7E-4	7.5E-4
Pb-210	1.1E-4	2.4E-4	4.0E-4	4.8E-4	5.1E-4	5.5E-4
Po-210	1.1E-4	2.5E-4	4.0E-4	4.8E-4	5.3E-4	5.5E-4
Lake Athabasca						
U-238	2.3E-2	4.9E-2	7.4E-2	7.9E-2	7.7E-2	7.5E-2
U-234	2.4E-2	5.0E-3	7.5E-2	8.1E-2	7.9E-2	7.8E-2
Th-230	5.3E-5	1.1E-4	1.7E-4	1.8E-4	1.8E-4	1.8E-4
Ra-226	3.1E-4	6.4E-3	9.7E-4	1.0E-3	1.0E-3	9.9E-4
Pb-210	1.3E-4	3.0E-4	4.9E-4	6.0E-4	6.6E-4	7.1E-4
Po-210	1.5E-4	3.2E-4	5.1E-4	6.0E-4	6.7E-4	7.1E-4

concentrations in both water and sediments are predicted to be statistically indistinguishable from background.

D.1.2 External Doses to Humans. Radionuclides released to surface waters will lead to human exposure through ingestion of water and food, and from direct exposure to beach sediments. The external exposure from sediments is obtained for each radionuclide as the product of sediment concentration, an exposure factor and an external dose conversion factor (ICRP 1977). The exposure factor represents the fraction of time spent on beaches. Adult members of the critical group, destined to receive the greatest exposure, were estimated to spend 1.3, 0.6, and 0.37% of their time swimming, walking, and sunning on beaches. For regional populations downstream of the critical group, the corresponding exposure factors were assumed to be 0.334, 0.063, and 0.032%. Only 32% of the people in these populations were considered to engage in swimming and 12% to use the beaches for walking and sunning. External exposure of infants (<1 year old) was by bathing only and infant dose calculations were restricted to the critical group. External dose conversion factors for each radionuclide are listed in Table D.3 and Tables 14 and 15. They include dose contributions from both parent and daughter radionuclides. Contributions from the beach are based on air gamma dose calculations one metre above the sediment (Beck et al. 1968). Contributions from water are based on calculations by Kocher (1979) and Kocher and Eckerman (1981). All values are converted to an effective dose equivalent (CEC 1979). The skin dose from contaminated water represents the sum of photon and electron doses 70 μm below the skin surface. The estimated doses for aquatic exposure pathways are presented in Tables 14 and 15.

TABLE D.3 DOSE CONVERSION FACTORS FOR EXTERNAL EXPOSURE

	Sediment Dose Factors (Sv/a per Bq/kg)		Water Dose Factors (Sv/a per Bq/L)		
Parent Radionuclide	Individual	Collective	Individual	Collective	Skin
U-238[1]	3.0E-8	3.8E-8	1.2E-7	1.2E-7	2.3E-7
U-234	8.1E-10	1.0E-9	6.2E-9	7.8E-9	7.6E-9
Th-230	8.6E-10	1.1E-9	5.9E-9	7.6E-9	7.3E-9
Ra-226[2]	3.2E-6	4.1E-6	7.0E-6	8.9E-6	1.1E-5
Pb-210[3]	3.2E-9	4.1E-9	2.6E-8	3.2E-8	7.8E-7
Po-210	1.6E-11	2.0E-11	3.5E-11	4.3E-11	4.3E-11

[1] U-238 factors include contributions from Th-234 and Pa-234.

[2] Ra-226 includes Rn-222, Pb-214, Bi-214, Po-214, Pb-210, Po-210 in air and Rn-222, Po-218, Pb-214, Bi-214, Po-214 in water.

[3] Pb-210 includes Po-210 in air, Bi-210 and Po-210 in water.

TABLE D.4 DOSE CONVERSION FACTORS FOR INTERNAL EXPOSURE (Sv/Bq)

Radionuclide	Infant	Adult Individual	Collective
U-238	1.3E-7	6.2E-8	6.2E-8
U-234	1.4E-7	7.0E-8	7.0E-8
Th-230	5.4E-5	1.5E-7	1.5E-7
Ra-226	1.2E-6	3.0E-7	3.0E-7
Pb-210	7.0E-1	1.4E-6	1.4E-6
Po-210	8.1E-1	4.3E-7	4.3E-7

TABLE D.5 INGESTION RATES FOR WATER AND AQUATIC FOODS

	Infant (0-1a)	Adult
Drinking Water (L/a)	300	800
Fish (kg/a)	2	44
Wild Rice (kg/a)	0	1

TABLE D.6 ANNUAL HARVEST BY COMMERCIAL AND SPORT FISHERIES

Lake	Commercial Catch 1975-79 (kg/a)	Sport Catch 1975-76 (kg/a)
Wollaston	230 234	23 088
Reindeer	198 442	521
Hatchet	-	18 620
Black	15 895	-
Athabasca	330 000	-

External dose rates to individuals of the critical group after 15 years of tailings discharge are given in Table D.8. Maximum annual collective dose commitments and 30-year collective dose commitments to regional populations are given in Tables D.9 and D.10.

TABLE D.7 POPULATION STATISTICS FOR COMMUNITIES NEAR CONTAMINATED WATER

Lake	Residents*			Transients**
	Infant	Adult	Total	
Wollaston	17	460	476	92
Wollaston Lake	2	157	159	
Lac la Hache	15	303	317	
Reindeer	92	2 487	2 579	147
South End	14	303	317	
Ballantyne Band	78	2 184	2 262	
Black	40	929	969	66
Stoney Rapids	7	242	249	
Stoney R Band	33	687	720	
Athabasca	70	3 021	4 035	-
Uranium City	49	1 940	1 989	
Eldorado	3	385	388	
Camsell Portage	0	11	11	
Fond du Lac	0	21	21	
Fond du Lac Band	18	664	682	
Fort Chipewyan	-	-	944	

* Data from the Saskatchewan Hospital Services Plan 1982.
** Data from the Midwest Lake Environmental Baseline Study (Beak 1980).

D.1.3 Internal Doses from Water and Food. Ingested radionuclides are not evenly distributed throughout the body. They tend to be localized in particular tissues with unique radiation sensitivities. Moreover, different radionuclides emit different radiations, some of which are more damaging than others. The consequences of internal irradiation are best evaluated in terms of an 'effective dose equivalent' (H_E) representing the whole-body dose of standard radiation type which would produce an equivalent health risk:

$$H_E = \sum_T W_T H_T$$

where H_T = the dose equivalent of standard radiation in tissue T.

TABLE D.8 MAXIMUM DOSE RATES TO CRITICAL GROUP INDIVIDUALS (mSv/a)

	Water	Rice	Fish	Swimming	Sunning	Total
Wollaston Lake Infants						
U-238	0.405E-5		0.297E-6	0.135E-9		0.435E-5
U-234	0.451E-5		0.331E-6	0.865E-11		0.484E-5
Th-230	0.126E-4		0.277E-5	0.605E-13		0.154E-4
Ra-226	0.974E-6		0.357E-6	0.252E-9		0.133E-5
Pb-210	0.223E-5		0.163E-5	0.363E-12		0.386E-5
Po-210	0.236E-6		0.866E-7	0.450E-13		0.323E-6
	0.246E-4		0.547E-5	0.144E-9		0.301E-4
Wollaston Lake Adults						
U-238	0.517E-5	0.130E-6	0.285E-5	0.135E-9	0.308E-8	0.815E-5
U-234	0.602E-5	0.150E-6	0.331E-5	0.865E-11	0.866E-10	0.948E-5
Th-230	0.905E-7	0.340E-8	0.149E-6	0.605E-13	0.677E-12	0.243E-6
Ra-226	0.650E-6	0.406E-7	0.179E-5	0.252E-9	0.878E-8	0.248E-5
Pb-210	0.114E-5	0.571E-8	0.627E-5	0.363E-12	0.602E-11	0.742E-5
Po-210	0.336E-6	0.168E-7	0.923E-6	0.450E-15	0.305E-13	0.126E-5
	0.134E-4	0.347E-6	0.153E-4	0.396E-9	0.120E-7	0.290E-4
Lake Athabasca Infants						
U-238	0.327E-6		0.240E-7	0.109E-10		0.351E-6
U-234	0.364E-6		0.267E-7	0.698E-12		0.391E-6
Th-230	0.303E-6		0.666E-7	0.146E-14		0.370E-6
Ra-226	0.399E-7		0.146E-7	0.103E-10		0.545E-7
Pb-210	0.830E-7		0.609E-7	0.136E-13		0.144E-6
Po-210	0.161E-7		0.591E-8	0.307E-16		0.220E-7
	0.113E-5		0.199E-6	0.219E-10		0.133E-5
Lake Athabasca Adults						
U-238	0.418E-6	0.104E-7	0.230E-6	0.109E-10	0.249E-9	0.658E-6
U-234	0.519E-6	0.122E-7	0.227E-6	0.698E-12	0.698E-11	0.758E-6
Th-230	0.218E-8	0.817E-10	0.359E-8	0.146E-14	0.166E-13	0.585E-8
Ra-226	0.266E-7	0.166E-8	0.731E-7	0.103E-10	0.358E-9	0.101E-6
Pb-210	0.426E-7	0.213R-9	0.234E-6	0.136E-13	0.240E-12	0.277E-6
Po-210	0.229E-7	0.115E-8	0.631E-7	0.307E-16	0.122E-14	0.872E-7
	0.103E-5	0.257E-7	0.831E-6	0.219E-10	0.601E-9	0.189E-5

TABLE D.9 MAXIMUM ANNUAL COLLECTIVE DOSE COMMITMENT TO REGIONAL POPULATIONS (Sv/a)

	Water	Rice	Fish	Swimming	Sunning	Total
Russell - Wollaston Lake						
U-238	0.189E-4	0.472E-6	0.252E-5	0.157E-9	0.130E-8	0.219E-4
U-234	0.220E-4	0.548E-6	0.293E-5	0.102E-9	0.363E-10	0.255E-4
Th-230	0.163E-6	0.611E-8	0.105E-6	0.353E-13	0.140E-12	0.274E-6
Ra-226	0.282E-5	0.176E-6	0.196E-5	0.277E-9	0.339E-8	0.496E-5
Pb-210	0.332E-5	0.166E-7	0.509E-5	0.332E-13	0.223E-11	0.843E-5
Po-210	0.634E-6	0.316E-7	0.655E-6	0.264E-15	0.113E-13	0.132E-5
	0.478E-4	0.125E-5	0.133E-4	0.536E-9	0.473E-8	0.624E-4
Reindeer Lake						
U-238	0.419E-6	0.109E-7	0.222E-6	0.350E-10	0.287E-10	0.652E-6
U-234	0.486E-6	0.126E-7	0.258E-6	0.227E-11	0.802E-12	0.757E-6
Th-230	0.732E-8	0.293E-9	0.112E-7	0.159E-14	0.619E-14	0.188E-7
Ra-226	0.671E-7	0.437E-8	0.177E-6	0.662E-11	0.806E-10	0.248E-6
Pb-210	0.949E-7	0.497E-9	0.497E-6	0.953E-14	0.556E-13	0.597E-6
Po-210	0.291E-7	0.152E-8	0.759E-7	0.121E-16	0.281E-15	0.107E-6
	0.110E-5	0.302E-7	0.124E-5	0.124E-10	0.110E-9	0.237E-5
Hatchet - Black Lake						
U-238	0.130E-7	0.119E-8	0.457E-7	0.109E-12	0.895E-12	0.599E-7
U-234	0.151E-7	0.138E-8	0.527E-7	0.704E-13	0.250E-13	0.692E-7
Th-230	0.227E-9	0.312E-10	0.267E-8	0.494E-16	0.193E-15	0.293E-8
Ra-226	0.209E-8	0.477E-9	0.369E-7	0.206E-12	0.251E-11	0.395E-7
Pb-210	0.298E-8	0.539E-10	0.107E-6	0.298E-15	0.174E-14	0.110E-6
Po-210	0.914E-9	0.165E-9	0.165E-7	0.383E-18	0.880E-17	0.176E-7
	0.343E-7	0.330E-8	0.261E-6	0.386E-12	0.343E-11	0.299E-6
Lake Athabasca						
U-238	0.169E-5	0.442E-7	0.931E-6	0.141E-10	0.116E-9	0.267E-5
U-234	0.197E-5	0.491E-7	0.108E-5	0.914E-11	0.324E-11	0.310E-5
Th-230	0.881E-8	0.331E-9	0.145E-7	0.192E-14	0.762E-14	0.236E-7
Ra-226	0.137E-6	0.856E-8	0.378E-6	0.135E-10	0.164E-9	0.524E-6
Pb-210	0.172E-6	0.862E-9	0.949E-6	0.172E-13	0.110E-12	0.112E-5
Po-210	0.927E-7	0.464E-8	0.255E-6	0.387E-16	0.557E-15	0.352E-6
	0.407E-5	0.108E-6	0.361E-5	0.368E-10	0.283E-9	0.779E-5

TABLE D.10 COLLECTIVE DOSES TO REGIONAL POPULATIONS (Person-Sv)*

	Water	Rice	Fish	Swimming	Sunning	Total
Russell - Wollaston Lake						
U-238	0.302E-3	0.753E-5	0.403E-4	0.252E-8	0.287E-7	0.350E-3
U-234	0.351E-3	0.876E-5	0.468E-4	0.163E-8	0.804E-9	0.407E-3
Th-230	0.261E-5	0.978E-7	0.168E-5	0.564E-12	0.311E-11	0.439E-5
Ra-226	0.451E-4	0.281E-5	0.314E-4	0.442E-8	0.749E-7	0.793E-4
Pb-210	0.531E-4	0.265E-6	0.814E-4	0.531E-11	0.415E-10	0.135E-3
Po-210	0.102E-4	0.506E-6	0.104E-4	0.442E-14	0.201E-12	0.211E-4
	0.764E-3	0.200E-4	0.212E-3	0.858E-8	0.104E-6	0.997E-3
Reindeer Lake						
U-238	0.668E-5	0.173E-7	0.354E-5	0.559E-10	0.624E-9	0.102E-4
U-234	0.777E-5	0.202E-6	0.412E-5	0.363E-10	0.175E-10	0.121E-4
Th-230	0.117E-6	0.468E-8	0.180E-6	0.254E-13	0.135E-12	0.302E-6
Ra-226	0.107E-5	0.699E-7	0.284E-5	0.106E-9	0.175E-8	0.398E-5
Pb-210	0.152E-5	0.795E-8	0.797E-5	0.152E-12	0.105E-11	0.950E-5
Po-210	0.464E-6	0.243E-7	0.121E-5	0.195E-15	0.535E-14	0.170E-5
	0.176E-4	0.326E-6	0.199E-4	0.198E-10	0.239E-8	0.378E-4
Hatchet - Black Lake						
U-238	0.208E-6	0.190E-7	0.725E-6	0.173E-11	0.195E-10	0.952E-6
U-234	0.242E-6	0.221E-7	0.842E-6	0.113E-11	0.544E-12	0.111E-5
Th-230	0.365E-8	0.499E-9	0.426E-7	0.789E-15	0.421E-14	0.467E-7
Ra-226	0.335E-7	0.763E-8	0.591E-6	0.329E-11	0.547E-11	0.632E-6
Pb-210	0.476E-7	0.861E-9	0.171E-5	0.476E-14	0.328E-13	0.176E-5
Po-210	0.146E-7	0.263E-8	0.264E-6	0.612E-17	0.168E-15	0.281E-6
	0.549E-6	0.527E-7	0.417E-5	0.616E-11	0.256E-10	0.478E-5
Lake Athabasca						
U-238	0.270E-4	0.676E-6	0.149E-4	0.226E-9	0.254E-8	0.426E-4
U-234	0.313E-4	0.786E-6	0.173E-4	0.146E-9	0.708E-10	0.494E-4
Th-230	0.141E-6	0.529E-8	0.234E-6	0.306E-13	0.167E-12	0.380E-6
Ra-226	0.219E-5	0.137E-6	0.603E-5	0.215E-9	0.359E-8	0.836E-5
Pb-210	0.276E-5	0.138E-7	0.152E-4	0.277E-12	0.204E-11	0.180E-4
Po-210	0.148E-5	0.742E-7	0.409E-5	0.620E-15	0.105E-13	0.564E-5
	0.649E-4	0.169E-5	0.578E-4	0.587E-9	0.620E-8	0.124E-3

* From 15 years of operation and 15 years following close-out.

TABLE D.11 FACTORS EMPLOYED IN DETERMINING DOSES TO AQUATIC BIOTA

Radionuclide	E_α (MeV)	E_β (MeV)	Γ (rad/h per mCi/cm^2)
Uranium*	9	0.87	0.074
Th-230	4.7	-	-
Ra-226	4.8	-	0.03
Rn-222 (+ short-lived daughters)	24.0	0.75	5.1
Pb-210**	5.3	0.41	0.02
Po-210	5.3	-	-

* Includes U-238, Th-234, Pa-234, U-234.

** Includes Bi-210, Po-210.

TABLE D.12 BIOACCUMULATION FACTORS FOR AQUATIC BIOTA (Bq/g per Bq/ml)

Radionuclide	Phytoplankton[2]	Invertebrates[3]	Fish[4]	Fish Eggs[5]
Uranium[1]	1 000	100	10	10
Th-230	1 500	500	150	100
Ra-226	2 500	1 000	100	100
Pb-210	200	100	275	10
Po-210	2 000	1 000	640	10

[1] Includes U-238, Th-234, Pa-234, U-234.

[2] Factors based on Thompson et al., 1972; US NRC, 1977.

[3] Factors based on Thompson et al., 1972; US NRC, 1977; Anderson et al., 1961.

[4] Factors based on Thompson et al., IAEA, 1976; Pradel and Zettwoog, 1977.

[5] Factors based on Polikarpov, 1966; Till et al., 1976; Nishiwaki et al., 1979.

TABLE D.13 DOSE RATES TO BIOTA FROM INCORPORATED RADIONUCLIDES (μGy/h per Bq/g)

Radionuclide	Plankton	Crayfish	Clams	Fish
Uranium*				
Alpha	1.6	0.5E+1	0.5E+1	0.5E+1
Beta	0.4E-2	0.5	0.5	0.5
Gamma	0.5E-6	2.1E-4	0.9E-4	0.8E-3
Th-230				
Alpha	0.8	0.3E+1	0.3E+1	0.3E+1
Ra-226				
Alpha	0.8	0.3E+1	0.3E+1	0.3E+1
Gamma	2.0E-7	0.9E-4	0.4E-4	0.3E-3
Pb-210 (+ Daughters)				
Alpha	0.9	0.3E+1	0.3E+1	0.3E+1
Beta	0.5E-2	2.6E-1	2.6E-1	2.6E-1
Gamma	1.3E-7	0.6E-4	2.6E-5	2.2E-4
Po-210				
Alpha	0.9	0.3E+1	0.3E+1	0.3E+1

* Includes U-238, Th-234, Pa-234, U-234.

W_T is the weighting factor for tissue T, giving the stochastic risk from tissue irradiation relative to that from whole-body irradiation at the same dose. Weighting factors have been defined in ICRP 26 (1977) for all major body tissues and organs except skin.

The 'collective effective dose equvalent' to the population as a whole is given by:

$$S_E = \int_0^\infty H_E N(H_E)\, dH_E$$

where $N(H_E)$ = the number of individuals receiving an effective dose equivalent in the range H_E to $H_E + dH_E$.

TABLE D.14 DOSE RATES TO BIOTA FROM WATER (µGy/h per Bq/L)

Radionuclide	Plankton	Crayfish	Clams	Fish
Uranium*				
Alpha	0.4E-2	-	-	-
Beta	0.5E-3	-	-	-
Gamma	2.1E-6	2.1E-6	2.1E-6	2.1E-6
Th-230				
Alpha	1.9E-3	-	-	-
Ra-226				
Alpha	1.9E-3	-	-	-
Gamma	0.3E-5	0.3E-5	0.3E-5	0.3E-5
Rn-222 (+ Short-lived Daughters)				
Alpha	0.8E-2	-	-	-
Beta	0.4E-3	-	-	-
Gamma	0.5E-3	0.5E-3	0.5E-3	0.5E-3
Pb-210 (+ Daughters)				
Alpha	2.1E-3	-	-	-
Beta	2.3E-4	-	-	-
Gamma	1.1E-3	1.1E-3	1.1E-3	1.1E-3
Po-210				
Alpha	2.1E-3			

* Includes U-238, Th-234, Pa-234, U-234.

The 'collective effective dose equivalent commitment' (S_E^C) is obtained by integrating S_E over time:

$$S_E^C = \int_0^\infty S_E(t)\,dt$$

In practice, integration over a finite period of 50 years is standard. For internal exposures, S_E^C may be expressed more conveniently as:

$$S_E^C = \bar{H}_E \int_0^\infty I.C(t)\,dt$$

where $\bar{H}_E$ = the population-averaged effective dose equivalent per unit of ingested activity, approximated by internal dose conversion factors from ICRP 30 (1979) for adults and from Johnson and Dunford (1983) for infants (Tables D.3 and D.4). They include contributions from parent and daughter radionuclides in equilibrium over the 50-year integration period.

I = the ingestion rate of the population (kg/a).

$C(t)$ = the radionuclide concentration in the ingested food or water at time t (Bq/kg).

The total collective effective dose equivalent commitment (S_y) incorporates two additional terms to account for hereditary effects on subsequent generations, and for fatal skin cancers:

$$S_y = S_E^C + 0.25\, S^C\text{gonads} + 0.01\, S^C\text{skin}$$

Gonadal dose conversion factors were available only for Ra-226.

Ingestion rates for contaminated water and food are listed in Table D.5. Infant ingestion was used only in the critical group dose calculations. Fish and drinking water are the major aquatic pathways for internal radionuclide exposure in northern Saskatchewan. The wild rice harvest is limited since the most productive areas are further south. However, rice harvesting permits are issued on Reindeer Lake and Lake Athabasca. Data from the Department of Northern Saskatchewan Economic Development Branch indicate that there may be as many as 15 permits for harvest on or near downstream waters. At an average yield of 381 kg/a/permit, annual production could approach 6000 kg/a. In fact, only 500 kg were harvested in 1982 and this was all used for seeding. For modelling purposes, an individual consumption rate of 1 kg/a, based on distribution of the maximum potential harvest over the entire population of the downstream water system, was assumed. This procedure probably overestimates actual consumption and allows for possible expansion of the wild rice industry.

Much of the fish harvest (Table D.6) is exported. Catch statistics are from the Midwest Lake Environmental Baseline Study (Beak 1980). Doses to critical group populations are based on an average consumption rate of 44 kg/a, all of which is assumed to be locally caught. Regional dose commitments are based on the entire catch from commercial and sport fisheries, wherever the catch exceeds local consumption. This procedure assumes that all surplus is exported and eventually eaten.

Population statistics used to calculate population dose commitments (Table D.7) are based on the Department of Northern Saskatchewan, Saskatchewan Hospital Services Plan for 1982. The Fort Chipewayan population is from the 1981 Census, Statistics Canada. Transient population estimates are based on fishing camp capacity from the Midwest Lake Environmental Baseline Study (Beak 1980) and a peak occupancy rate of 78%.

Internal dose rates to individuals of the critical group after 15 years of tailings discharge are given in Table D.8. Maximum annual collective dose commitments, and 30-year collective dose commitments to regional populations are given in Table D.9 and D.10.

Non-stochastic dose response was not considered in this study as individual dose rates within the critical group were well below levels where non-stochastic responses would be expected.

D.1.4 **Critical Group and Annual Collective Doses.** Maximum dose rates to critical group individuals (Table D.8) occur between 5 and 15 years after the start of mining operations. This period coincides with the period of peak radionuclide concentrations in water (Tables D1 and D2). The major pathway for human exposure is by ingestion of radionuclides, either directly from water by drinking, or by consumption of wild rice and fish from the water. External exposure by swimming, or walking and sunning on beaches, is at least three orders of magnitude lower than internal exposure by ingestion. The beach component dominates the external exposure pathway, reaching its maximum at the time of peak sediment activity. For uranium, thorium, and radium, this occurs 18 years after the start of mining operations.

Adult dose rates generally exceed those of infants due to the greater adult intake of contaminated food and water (Table D.5). However, this is not true for all radionuclides. Thorium produces a higher dose rate in infants than adults. This is due to the high dose conversion factor for thorium in infants. High infant dose conversion factors are expected for bone-specific radionuclides as a result of rapid bone growth in infants as opposed to adults. However, the three order of magnitude difference indicated in Tables D.3 and D.4 is probably excessive, and due in part to the fact that infant and adult dose conversion factors were taken from different sources. Infant factors are from Johnson and Dunford (1983). Even their adult factors for thorium are much greater than the ICRP values for adults which were used in this model. Infant dose conversion factors are not available from ICRP.

Dose rate contributions from particular radionuclides are not in strict proportion to the concentrations of those radionuclides in water. This is because food chain transfer coefficients and dose conversion factors differ greatly among radionuclides. Both tend to be greater for uranium daughter nuclides than for uranium itself (Tables B.4, D.3, and D.4). Thus, dose rates from lead in fish exceed those from uranium, even though uranium concentrations in water are two orders of magnitude higher than those of lead.

Critical group calculations are shown in Table D.8 for Lake Athabasca residents as well as for those of Wollaston Lake. The comparison indicates that Wollaston Lake dose rates from the Key Lake and Rabbit Lake sites are approximately an order of magnitude higher than those at Lake Athabasca, contributed mainly from the Cluff Lake site. This is in agreement with the lower radionuclide concentrations in Lake Athabasca, and the lower output of the Cluff Lake site. Even at these low levels, the calculated incremental dose rates for Athabasca residents are probably higher than actually received because of the mixing assumptions used for the waters of Lake Athabasca. The incremental dose rates to the residents of Lake Athabasca are in the order of 10^{-4}% of natural background.

The maximum annual dose commitments to regional populations are shown in Table D.9. These reflect population size, as well as radionuclide distribution and individual consumption rates. The Russell-Wollaston region includes a transient population on Otter Bay, estimated at 25 individuals, as well as the Wollaston Lake community. It also includes all individuals destined to consume fish exported from the region. Thus, the calculated annual collective dose commitment for the region exceeds that which might be calculated for the Wollaston Lake community alone.

Fish exports are also included in the calculated annual collective dose commitment for the Reindeer Lake, Hatchet to Black Lake, and Lake Athabasca regions. Annual collective dose commitments within communities decrease with downstream distance from a mine site. The annual collective dose commitment to the Russell-Wollaston Lake region exceeds that to the Hatchet-Black Lake region by two orders of magnitude, and that to the Lake Athabasca region by one order of magnitude.

D.1.5 Thirty-year Collective Dose Commitments. Collective dose commitments to regional populations integrated over the 30-year time frame of the study (Table D.10) are approximately 16 times greater than the maximum annual collective dose commitment.

Near-maximum annual collective dose commitments were maintained for the duration of the 15-year operational period. Thus, the post-operational period accounts for a small fraction (about 3%) of the collective dose commitment. The annual collective dose commitment in Year 30 accounts for approximately 10^{-3}% of the 30-year collective dose commitment and can be considered negligible relative to the calculated incremental collective dose commitment received during the operational period of the mines. Within a broader context, the overall incremental collective dose commitment contributed as a result of the mining and milling releases to aquatic pathways is less than one million of that received from natural background sources.

D.2 DOSE RATES TO AQUATIC BIOTA

Aquatic biota are important for their economic value, as well as for the part they play in radionuclide transfer through the food chain to man. For this reason, dose rates to aquatic biota are calculated. An extensive literature on dose and dose rate effects in aquatic organisms has been reviewed in NRCC 19250 (1983), IAEA TR-190 (1979), Blaylock and Trabalka (1978), IAEA TR-172 (1976), Andrushaytis (1973), Chipman (1972), Il'yenko (1969) and Polikarpov (1966). Dose rate calculations are based on IAEA TR-172 and Woodhead (1970). All dose rates to aquatic biota are expressed in Grays/unit time. This is because there are no appropriate quality factors for radiation that are applicable to species other than man.

D.2.1 Dose Rates from Incorporated Radionuclides. Internal dose rates DR(Gy/hr) from alpha (α), beta (β), and gamma (γ) radiation are calculated on the assumption of uniform radionuclide distribution through the organism:

$$DR(\alpha) = N_\alpha (5.8 \times 10^{-4}) E_\alpha C$$

$$DR(\beta) = N_\beta (5.8 \times 10^{-4}) E_\beta C$$

$$DR(\gamma) = 2.7 \times 10^{-7} \Gamma p \bar{g} C$$

where N_α and N_β = constants equal to unity for all large organisms (crayfish, clams, fish). For plankton, $N_\alpha = 0.3$ and $N_\beta = 0.007$.

E_α and E_β = radionuclide emission energies given in the Radiological Health Handbook (1970) and Table D.11.

Γ = the gamma ray constant for each radionuclide (Table D.12).

p = organism density (1 g/cm^3).

$\bar{g}$ = a mean geometric factor representing the size and shape of the organism.

C = the radionuclide concentration in the tissue of the organism (Bq/kg), calculated as the product of water concentration and a bioaccumulation factor.

Plankton are represented as spheres 50 microns in diameter ($\bar{g} = 0.02$). In northern Saskatchewan lakes, this model is appropriate for zooplankton and on the conservative side for phytoplankton. Crayfish are represented as cylinders 2 cm in radius and 5 cm long ($\bar{g} = 10.6$). Clams are represented as spheres 1 cm in radius ($\bar{g} = 4.7$), a model appropriate to the common fingernail clams of the region. Fish are represented as cylinders 10 cm in radius and 50 cm long ($\bar{g} = 41$).

Bioaccumulation factors vary widely with environmental conditions, and ideally, site-specific values should be obtained.

The values used for dose calculations (Tables 19 and D.12 are representative of those reported in the literature. Bioaccumulation factors calculated from radionuclide concentrations in northern Saskatchewan fish (Environment Canada, 1978) tend to fall in the lower range of values taken from the literature. Dose rates from incorporated radionuclides are calculated in Table D.13 assuming a tissue concentration of 0.037 Bq/g. Actual dose rates to biota exposed to the maximum radionuclide concentrations calculated in this study for Otter Bay are given in Table S.4.

D.2.2 Dose Rates from Water. External dose rates (µGy/hr) from alpha (α) irradiation by radionuclides in the water are calculated as in Section D.2.1 with $N\alpha = 0.7$ for plankton. For larger organisms, $N\alpha = 0$.

The beta (β) component of the external water dose rate for plankton is:

$$DR(\beta) = (1-N_\beta)(5.8 \times 10^{-4}) E_\beta C$$

All parameters are defined as in D.2.1. For larger organisms,

$$DR(\beta) = DR(\gamma) = 0.$$

The gamma (γ) component of the external water dose rate for any organism is given by:

$$DR(\gamma) = (2.7 \times 10^{-7}) E_\gamma C$$

where E_γ is the radionuclide emission energy (Radiological Health Handbook 1970).

External dose rates from water are calculated in Table D.14 assuming a water concentration of 0.037 Bq/L. Actual dose rates to biota within Otter Bay are given in Table S.4.

D.2.3 Dose Rates from Contaminated Sediments. The gamma dose rate in air (µrad/h) from uranium (in equilibrium with its daughters) in soil is given by Gustafson and Brar (1963) as:

$$Du = 0.779 \times 10^4 \, Su$$

where Su is uranium concentration (g of uranium per g of soil). The dose rate Du is essentially constant to about 1 m above ground level (Beck 1972). Du is 0.62 µGy/h per Bq/g.

The dose rate from gamma radiation is approximately proportional to the total emitted gamma ray energy (Rockwell 1956) in the energy range between 70 keV and 10

MeV. Consequently, the dose rate from particular radionuclides or groups of radionuclides may be calculated from the fraction of total gamma energy emitted by uranium in equilibrium with the daughters:

$$U^{238} - U^{234} \qquad DR(\gamma) = \frac{0.0315}{0.9589}(0.62) = 0.020\ \mu Gy/h$$

$$Ra^{226} \qquad DR(\gamma) = \frac{0.0057}{0.9589}(0.62) = 0.0037\ \mu Gy/h$$

Gamma dose rates from sediments are listed in Table D.15.

TABLE D.15 DOSE RATES TO BIOTA IN AND ON SEDIMENTS (μGy/h per Bq/g)

Radionuclides	Beta	Gamma
Uranium*	0.3	2.1E-2
Ra-226	-	3.8E-3
Short-lived Rn Daughters	0.21	0.6
Long-lived Rn Daughters	0.04	-

* Includes U-238, Th-234, Pa-234, U-234.

Beta dose rate calculations were based on a simplified sediment model of radioactivity concentrated in four infinite planes 1.25, 3.75, 6.25 and 8.75 mm below the surface. The sediment was assumed to have a bulk density of 1.5 g/cm^3. At 0.037 Bq, sediment contamination in each plane is 1.4 x 10^{-2} Bq.

Cross (1967) gives the dose rate in water above an infinite plane for a variety of beta-emitting radionuclides. These can be used as reference nuclides to derive dose rates for uranium series nuclides with similar beta energies (Table D.16).

The reference nuclide dose rate is multiplied by the beta percentage intensity of the uranium series nuclide and the contamination level in the sediment plane. The resulting beta dose rates as shown in Table D.15 for sediment concentrations of 0.037 Bq/g. Actual dose rates from sediments in Otter Bay during the period of peak sediment contamination are given in Table S.4.

D.2.4 Dose Rates to Fish Embryos. The fish egg model used to calculate embryonic dose rates is 2 mm in diameter, with an outer shell 20 μ thick. The embryo develops within 200 μ of the shell, at an average distance of 100 μ (Woodhead 1970). The alpha dose rate form incorporated radionuclides is calculated as in Appendix D.2.1 with $N_\alpha = 1$.

TABLE D.16 PARAMETERS FOR CALCULATING BETA DOSE RATES FROM SEDIMENT

Uranium Series Nuclide		Reference Nuclide		
Symbol	$E_\beta$max (Mev)	Symbol	$E_\beta$max (Mev)	Beta Dose Rate (Gy/h per Bq/cm^2)
Pa^{m}-234	2.3	Y-90	2.3	0.9
Pb-214	0.68	Cs-134	0.66	0.8E+2
Bi-214	1.0	Bi-210	1.2	1.2E-1
Bi-214	1.5	Y-91	1.5	0.5
Bi-214	3.3	K-42	3.5	1.5
T1-210	1.9	Ga-68	1.9	0.6
Bi-210	1.2	B-210	1.2	1.2E-1

Beta dose rates are based on the conservative calculations of Woodhead (1970) with radioactivity distributed uniformly in the egg.

Gamma dose rates in μGy/h were calculated from Woodhead's (1970) formulae for three radiation sources:

1. Activity adsorbed onto the egg shell surface:
   $DR(\gamma) = 20.5 \times 10^{-5}\Gamma$
2. Activity uniformly distributed throughout the egg:
   $DR(\gamma) = 0.82 \times 10^{-5}\Gamma$
3. Activity in surrounding water (Bq/kg):
   $DR(\gamma) = 5.8 \times 10^{-4} E_\gamma$

Parameters are defined as in D.2.1 (Radiological Health Handbook 1970), and Table D.11.

Dose rates to fish embryos at unit radionuclide concentrations in water and egg are shown in Table D.17. Actual dose rates to fish embryos in Otter Bay during the period of maximum contamination are given in Table S.4. Activity on the egg shell surface was assumed to be 10 times that taken up by the egg.

TABLE D.17 DOSE RATE TO FISH EMBRYOS (μGy/h)

		Type of Radiation		
Geometry	Radionuclide	Alpha	Beta	Gamma
Activity Incorporated into Egg (0.037 Bq/g)	Uranium*	1.9E-1	4.4E-3	6.1E-7
	Th-230	1.0E-1	-	-
	Ra-226	1.0E-1	-	2.5E-7
	Pb-210	1.1E-1	3.3E-3	1.6E-7
	Po-210	1.1E-1	-	-
Activity on Shell (0.037 Bq/cm^2)	Uranium	-	-	1.5E-5
	Ra-226	-	-	5.5E-6
	Pb-210	-	-	4.1E06
Activity in Water (0.037 Bq/L)	Uranium	-	1.6E-5	7.7E-8
	Ra-226	-	-	1.2E-7
	Rn-222 (+ Short-lived Daughters)	-	1.1E-5	2.0E-5
	Pb-210	-	4.8E-6	4.0E-5

* Includes U-238, Th-234, PA-234, U-234.